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The potential water quality effects and economic implications of soil and water conservation practices (SWCPs) are identified. Methods for estimating the effects of SWCPs on pollutant losses from croplands are presented. Mathematical simulation and linear programming models were used to estimate the effects of SWCPs on edge-of-field losses of sediment, nutrients, and pesticides and to evaluate the impacts on farm income of implementing SWCPs for pollution control. The models were applied to hypothetical farms and fields in New York, Iowa, Texas, and Georgia.

The major environmental benefit of SWCPs was determined to be their usefulness in controlling edge-of-field losses of sediment and total phosphorus from croplands. Farm plans for erosion control, however, are not necessarily the same as plans for sediment control. SWCPs may not be effective at reducing total losses of dissolved nitrogen although runoff losses will generally be reduced. These practices will also reduce losses of sediment-adsorbed pesticides but are not as effective as other techniques. Conservation tillage is often a cost-effective method of reducing losses of sediment, nutrients, and pesticides because it can be implemented with little or no decrease in crop yields.

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October 1979

EFFECTIVENESS OF SOIL AND WATER CONSERVATION PRACTICES FOR POLLUTION CONTROL

Edited

by

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Grant No. R804925010

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FOREWORD

As environmental controls become more costly to implement and the penalties of judgment errors become more severe, environmental quality management requires more efficient analytical tools based on greater knowledge of the environmental phenomena to be managed. As part of this Laboratory's research on the occurrence, movement, transformation, impact, and control of environmental contaminants, the Technology Development and Applications Branch develops management or engineering tools to help pollution control officials achieve water quality goals through watershed management.

Efforts to reach water quality goals include the identification and application of best management practices (BMPs) to control agriculturally related water pollutants. This report examines soil and water conservation practices used for many years in the United States to determine their potential as BMPs. Although these conservation practices are directed to erosion and water control rather than direct control of nonpoint source pollutants, their use in specific situations could provide environmental managers with an additional tool for controlling pollutants of potential concern.

David W. Duttweiler
Director
Environmental Research Laboratory
Athens, Georgia

ABSTRACT

The principal objectives of this project were to identify the potential water quality effects of soil and water conservation practices (SWCPs) and to describe the economic implications of their use. A secondary objective was to develop and test methodologies that could be used to estimate the effects of SWCPs on pollutant losses from croplands. The project was limited to non-irrigated field crops in the Eastern half of the United States. Potential water pollutants considered were sediment, nutrients, and pesticides. Project resources and duration did not permit the execution of experimental field studies. Rather, research objectives were accomplished using three related procedures. First, data from previous and ongoing field studies were collected and reviewed. Second, mathematical simulation and linear programming models were used to estimate the effects of SWCPs on edge-of-field losses of sediment, nutrients, and pesticides and to evaluate the impacts on farm income of implementing SWCPs for pollution control. The models were applied to hypothetical farms and fields in New York, Iowa, Texas, and Georgia. Third, a panal of outside experts from universities, government agencies, and consulting firms was convened three times during the study to review methods and results of analyses.

The major environmental benefit of SWCPs was determined to be their usefulness in controlling edge-of-field losses of sediment and total phosphorus from croplands. It was established, however, that farm plans for control of erosion are not necessarily equivalent to plans for sediment control. SWCPs may not be effective at reducing total losses (runoff plus percolation of dissolved nitrogen (nitrate) although runoff losses will generally be reduced. SWCPs implemented for erosion or sediment control will have additional benefits in terms of reducing losses of moderately or strongly adsorbed pesticides although in many cases alternative methods are more effective than SWCPs for control of pesticide losses. Conservation tillage will often be a cost-effective method of reducing losses of sediment, nutrients, and pestisides because it is a practice that can be implemented in many cases with little or no decrease in crop yields.

This report was submitted in fulfillment of Grant No. R804925010 by Cornell University under the sponsorship of the U.S. Environmental Protection Agency. This report covers the period October 25, 1976, to October 24, 1978. Work was completed as of January 31, 1979.

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SECTION 1

INTRODUCTION

Historically, water pollution abatement policies have focused on control of municipal and industrial sources. Interest in other potential sources has increased as the nation has expanded its water pollution concerns. The emphasis of national water pollution control policies now is on the amount of wastes that can be kept out of surface waters, rather than on the amount that can be assimilated by the waters. Areawide waste treatment management plans that are to be prepared as a result of Section 208 of the Federal Water Pollution Control Act Amendments of 1972 (PL 92-500) are to include identification of "agriculturally and silviculturally related nonpoint sources of pollution" and are to "set forth procedures and methods to control to the extent feasible such sources". Nonpoint sources will be difficult to control if control is defined as the ability to establish and enforce effluent standards. However, if the control is approached by using appropriate management practices, many of the sources are controllable. In the United States, the emphasis for control of nonpoint source contributions is being placed upon "best management practices" rather than on the collection, treatment, and effluent standards approach used for control of point sources.

Best management practices (BMPs) are defined as:

> "A practice or combination of practices that is determined by a designated areawide planning agency after problem assessment, examination of alternative practices and appropriate public participation to be the most effective, practicable (including technological, economic, and institutional considerations) means of preventing or reducing the amount of pollution generated by nonpoint sources to a level compatible with water quality goals."

The Clean Water Act of 1977 (PL 95-217) established a cost-sharing program whereby the Secretary of Agriculture, with the concurrence of the Administrator of the Environmental Protection Agency, may subsidize the installation of BMPs on farmlands for the purpose of water pollution control. This activity is known as the "Rural Clean Water Program" and is under the direction of the U.S. Soil Conservation Service. Successful implementation of the Rural Clean Water Program will depend on identification of suitable BMPs for control of agriculturally related water pollutants. This is not a simple task, for such pollutants are primarily associated with diffuse sources which are difficult to monitor and control.

Soil and water conservation practices (SWCPs) are agricultural practices that have the potential of being best management practices. However, the soil and water conservation practices utilized in the United States for over 150 years are designed for erosion and water control rather than for direct management of potential pollutants such as nutrients, oxygen demanding materials, or chemicals used for increased crop production or pest management.

Because SWCPs are utilized to retain the principal carriers of agricultural nonpoint sources (runoff water and eroded soil) on a field, there can be the tendency to consider all SWCPs as agricultural BMPs. This may not always be appropriate for reasons which include the following

1. SWCPs are effective at reducing soil erosion on cropland. However, many pollutants, including nitrogen, phosphorus and pesticides, also are lost from cropland in dissolved form in runoff and percolation. The effects of SWCPs on the transport of pollutants in solution is much less certain than their effects on erosion.

2. There are additional practices which are capable of reducing pollutant losses from cropland such as fertilizer management, integrated pest management and management of animal manure applications.

3. Water pollution from agricultural nonpoint sources results from a chain of circumstances. Pollutants are lost from a cropped field, transported through the landscape to surface or groundwater, and may or may not result in an objectionable problem (Figure 1-1). The relationship between any practice used on a field and surface or groundwater quality is uncertain.

This research project was designed to evaluate the effectiveness of SWCPs as potential BMPs and to clarify, where possible, the above uncertainties. Conclusions in this report regarding the effectiveness of SWCPs for nonpoint source pollution control are not reflections on their value for soil conservation. Prevention of cropland erosion has long been a national objective and the soil conservation benefits of SWCPs were not evaluated in this project.

DESCRIPTION OF THE PROJECT

Objective

The main objective of the project was to identify the potential water quality effects of SWCPs and to describe the economic implications of their use. The project evaluated the hypothesis that soil and water conservation can be an economical way of significantly reducing agricultural nonpoint source pollution. A secondary objective was to develop and test methodologies which could be used to estimate the effects of SWCPs on pollutant losses from arbitrary sites (cropped fields).

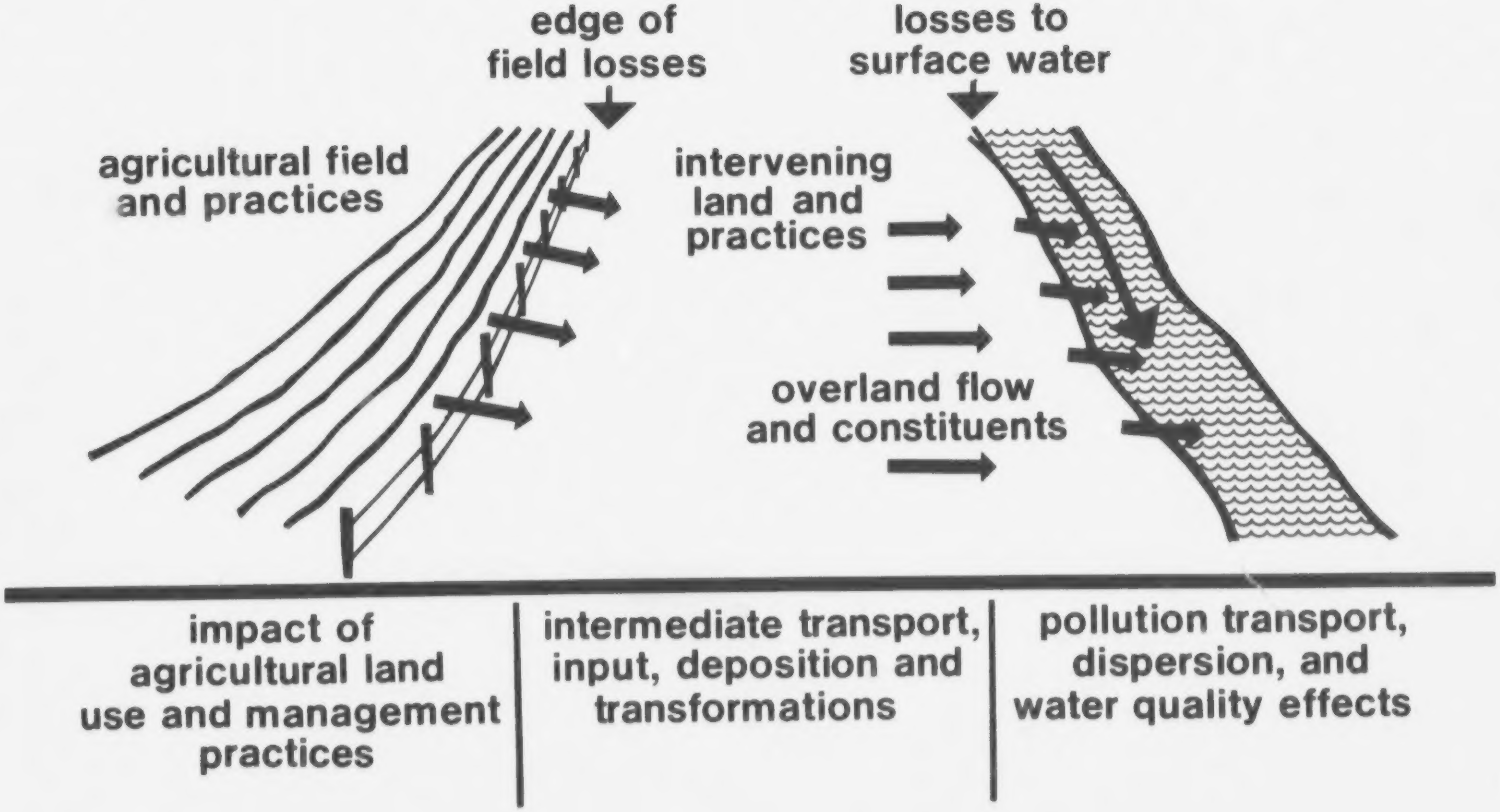
edge of field losses
losses to surface water
agricultural field and practices
intervening land and practices
overland flow and constituents
impact of agricultural land use and management practices
intermediate transport, input, deposition and transformations
pollution transport, dispersion, and water quality effects

FIGURE 1-1. TRANSPORT, TRANSFORMATION AND IMPACT OF POTENTIAL AGRICULTURAL NONPOINT SOURCES

Scope

The project had a two-year duration and was limited to non-irrigated field crops in the Eastern half of the United States. Project duration and resources did not permit the execution of experimental field studies. The potential water pollutants which were considered in the study were sediment, nutrients and pesticides. Effects of animal manures were not included. Project analyses were generally limited to edge-of-field (Figure 1-1) pollutant losses in surface runoff, subsurface water (interflow) and percolation. Where possible, implications for water quality management were explored. The project did not evaluate SWCPs in terms of any ultimate water quality benefits that might occur.

Approach

Research objectives were accomplished using three related procedures.

1. Data from previous and ongoing field studies were collected and reviewed to abstract information relevant to the effectiveness of SWCPs.

2. Mathematical simulation and linear programming models were used to estimate the effects of SWCPs on edge-of-field losses of sediment, nutrients and pesticides and to evaluate the economic impact on farms of implementing SWCPs for control of these pollutants. The models were used to study SWCPs on hypothetical farms and fields in New York, Iowa, Georgia and Texas.

3. A panel of outside experts (Table 1-1) was convened three times during the two-year study to review project direction, methods of analyses, regional differences in SWCP use and effectiveness, interpretation of information and results, and to review the conclusions.

The general methodology used to reach the conclusions could be of interest to individuals having responsibility for agriculturally related nonpoint source pollution control. It is recognized that each SWCP can control some of the pollutants of potential concern. However, the key to their use as a BMP is to understand the conditions under which they are effective and, for specific situations, to be able to determine their potential effectiveness. While the project results will be useful to interested individuals, many individuals and agencies will wish to reach their own conclusions of the effectiveness of SWCPs as BMPs in a site-specific situation. The approach used in this study to determine SWCP effectiveness may permit others to reach conclusions in their own situations.

PROJECT PARTICIPANTS

A multidisciplinary team of Cornell University faculty, staff and graduate students were involved in the study. The disciplines included were agricultural economics, agricultural engineering, agronomy, and environmental engineering. The individuals having major project responsibility have been

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listed as the authors and editors of the report. Many other individuals played key roles in the project and several are acknowledged in the specific chapters. We particularly would like to recognize the input of Nelson L. Bills and Fred N. Swader to the development of the project. Lee A. Mulkey, the project officer for the study, made many contributions of information, advice and encouragement which greatly facilitated work progress.

The project also had the benefit of assistance by Hydrocomp, Inc., Palo Alto, California. Hydrocomp utilized its capabilities to run the Agricultural Runoff Management (ARM) model to evaluate certain SWCPs. Appendix F presents the results of the Hyrocomp study. We wish to acknowledge and express appreciation for the participation of Hydrocomp, Inc.

We also greatly appreciate the interest, perception, positive criticism, and helpful input of the project's expert panel. The individuals who participated in the panel are identified in Table 1-1. Their advice was given freely throughout the project and was a significant contribution to the project. Arrangements for the panel meetings were handled by Colleen S. Martin in her usual competent fashion.

Last, but certainly not least, we wish to thank Doreen E. Kirchgraber for her many hours spent organizing meetings, typing and proofing reports, and carrying out countless other tasks which contributed to the successful completion of the project.

SECTION 2

CONCLUSIONS

The effects of soil and water conservation practices (SWCPs) on losses of agricultural chemicals from croplands in runoff and percolation are largely determined by the site-specific effects SWCPs have on sediment losses and water movement. Since most SWCPs significantly reduce sediment losses they also reduce losses of chemicals associated with sediment, such as organic nitrogen and phosphorus, inorganic particulate phosphorus and organochlorines. Most SWCPs reduce surface runoff water, but to a lesser extent than reductions in sediment. Reductions in runoff are usually associated with increases in subsurface drainage, most particularly percolation to groundwater aquifers.

Specific conclusions based on the comparative analysis of previous research and the results of mathematical simulation and linear programming models are summarized in the remainder of this section.

CONTROL OF SEDIMENT LOSSES

1. Cropland soil erosion control practices are generally used to hold soil in place. Sediment control practices are designed to prevent eroded soil from entering streams and lakes. Erosion control practices reduce sediment loss from fields. Cost effective sediment control requires consideration of practices which do not control erosion (e.g., sediment basins or stream buffer strips) and concentrating erosion control practices on fields with highest sediment delivery.

2. SWCPs are more effective in controlling sediment yield from fields than in reducing runoff.

3. SWCPs reduce field sediment yield by reducing soil detachment and its transport in runoff. Control of detachment by rainfall requires protection of the soil surface by residues or crop canopies. Soil detachment and transport by overland flow are controlled by reduction of the quantity and/or velocity of flow.

4. Adsorbed chemicals are associated with clay and organic fractions of sediment. These constituents are most effectively controlled by SWCPs which provide protection from raindrop splash, such as

sod-based rotations, no-tillage, and conservation tillage. SWCPs that rely on reducing sediment transport such as graded terraces or contouring are effective in controlling losses of organic matter and clay if these sediment fractions are in soil aggregates.

5. The potential for erosion is predictably variable from month to month dependent on local characteristic weather conditions and crop growth. Since the effectiveness of SWCPs is seasonally predictable, in some cases SWCPs can be matched with highest potential sediment loss periods so as to be most effective.

6. SWCPs that control erosion by reducing raindrop impact are less influenced by storm magnitude than practices which rely on control of soil detachment by flow or reduction of transport capacity. As the magnitude of runoff events increases, the effectiveness of some transport and flow detachment control type practices such as strip cropping or graded terraces decreases. Contouring can become ineffective under conditions of very high runoff.

CONTROL OF NUTRIENT LOSSES

7. Practices such as contouring, terraces, sod-based rotations, conservation tillage and no-tillage significantly reduce edge-of-field losses of solid-phase nitrogen and phosphorus because they reduce erosion.

8. Sod-based rotations significantly reduce losses of dissolved phosphorus and nitrogen in surface runoff.

9. Practices such as contouring, terraces, conservation tillage and no-tillage which reduce surface runoff reduce loss of dissolved nitrogen in surface runoff, although reductions are less than those of solid-phase nitrogen. These practices may increase dissolved nitrogen losses in subsurface drainage (percolation and/or interflow).

10. Practices such as no-tillage and conservation tillage which protect the soil surface with crop residues during the non-growing season have an uncertain effect on losses of dissolved phosphorus in surface runoff.

11. SWCPs such as contouring and terraces which are not based on residue management, decrease losses of dissolved phosphorus in surface runoff, although reductions are less than those of phosphorus associated with sediment.

12. Management of nitrogen fertilizer applications to meet crop needs can reduce the losses of dissolved nitrogen in both runoff and percolation.

13. Losses of all forms of nitrogen and phosphorus in runoff vary significantly from year to year. More importantly, the effects of SWCPs also show significant yearly variations.

CONTROL OF PESTICIDE LOSSES

14. Wider implementation of techniques which reduce the amount of pesticide used for crop protection offers an effective way of reducing the quantities of pesticide leaving crop fields. Cost-effective insect pest control programs are currently being developed which reduce insecticide use. Most of these programs integrate several pest control techniques and are generally described by the term "integrated pest management" (IPM). One of the most widely implemented IPM techniques consists of the use of population monitoring ("scouting") to determine when pesticide applications are necessary. More complex integrated pest management programs involving a combination of cultural and biological control methods have been implemented for cotton and are being developed for other crops.

15. Improper disposal of pesticide containers and dilute pesticide mixtures in applicaton tanks appear to be important sources of pesticide pollution. Legislative controls on pesticide disposal as well as technological advances in application equipment can effectively reduce these sources of pesticide pollution.

16. The seasonal losses of pesticides due to drift and volatilization are often greater than the amount of pesticide carried in runoff. Except for practices like sod-based rotations which reduce the amount of pesticide applied, SWCPs cannot usually reduce drift and volatilization. Drift losses can be reduced by changes in formulation or application procedures.

17. The effectiveness of SWCPs in controlling transport of five major pesticides was compared with the effectiveness of alternative practices. For the three insecticides studied (carbofuran, toxaphene, and methyl parathion) integrated pest management programs combining scouting with cultural pest control methods seem to be a more effective means of reducing the losses of insecticide in water and sediment than SWCPs. SWCPs are effective at reducing the quantity of paraquat in runoff. However, currently available information indicates that paraquat in runoff is irreversibly adsorbed to clay particles and does not appear to pose an environmental hazard since it is not available to biological organisms. Of the five pesticides considered, SWCP appeared to be among the most effective pollution control methods only for atrazine.

18. Most pesticides currently in use can be classified as moderately adsorbed. Total mass flux of such pesticides in runoff occurs primarily in the dissolved form. Thus SWCPs must reduce runoff water in order to control the losses of these pesticides.

19. Pesticides such as paraquat, toxaphene, DDT, and dieldrin which are strongly adsorbed to the soil are primarily associated with clay and organic matter. Runoff losses of these pesticides will be most effectively controlled by SWCPs which control erosion.

20. The modelling results indicate that generally the effect the SWCP on downward movement of pesticides below the root zone is minimal. However, field and experimental data to support this conclusion are lacking. In particular this conclusion would not hold for a persistent, nonadsorbed pesticide.

21. The length of time between a pesticide application and the first major runoff event following application has a very significant effect on the amount of pesticide carried in the runoff, especially for pesticides which are not persistent. Thus, improved timing of applications could significantly reduce loss of these pesticides.

COST-EFFECTIVENESS

22. Although both the cost and effectiveness of all SWCPs may vary considerably for different situations, conservation tillage methods appear to have widespread potential for reducing soil erosion and sediment delivery with little or no negative effect and sometimes a positive effect on net farm income. Thus in many cases, these practices should form the basis for farm plans designed to reduce soil erosion and/or to reduce sediment delivery.

23. The most economical set of practices for a farm may depend on whether the goal is to reduce sediment delivery to waterways or to reduce levels of soil erosion. In particular, farm plans designed primarily to control sediment delivery rather than soil erosion can often specify location of crops and practices relative to waterways and utilize sediment traps, thereby resulting in a reduction in sediment delivery which is proportionally greater than the reduction in soil erosion. Thus the cost of achieving a reduction in sediment delivery can be significantly less than the cost of an equivalent percentage reduction in soil erosion. The benefits of reducing both soil erosion and sediment delivery should be considered in the design of farm plans for improving water quality.

24. Although systems of practices designed to reduce soil erosion generally achieve a reduction in sediment delivery roughly proportional to the reduction in erosion, a few practices are notable exceptions. On fields which have a relatively low sediment delivery ratio (SDR), practices such as diversion ditches and sod waterways may, in some cases, increase the SDR, resulting in increased levels of sediment delivery. On fields with high SDRs, practices such as tile-outlet terrace systems may decrease the SDR due to sediment deposition within the terrace channel. Such practices will be relatively more efficient in reducing sediment delivery than in reducing soil erosion.

25. For any given farm, the marginal cost of reducing sediment delivery increases as required reductions become greater. To achieve relatively small reductions, practices such as conservation tillage, contouring, and strip-cropping (in areas where close-seeded crops are normally grown) can often be implemented with little change in farm income. To achieve greater reductions in sediment delivery, practices which

are generally more costly, such as widely spaced terraces, must be used. High levels of reduction require implementation of practices generally associated with sharply increased farm costs. Typical practices (usually in combinations) would include closely spaced terrace systems, substitution of sod crops for row crops, and removal of land from crop production. Implementation of such practices generally results in a markedly greater farm cost and a considerable reduction in the level of cost-effectiveness.

26. Farm plans designed to reduce soil erosion and/or sediment delivery will not necessarily reduce losses of dissolved nutrients.

27. Reducing levels of sediment delivery to a waterway will reduce levels of solid-phase nutrients reaching the waterway. The reductions, however, may be proportionally less than the reductions in sediment delivery in situations where the nutrient enrichment ratio increases as levels of sediment delivery are decreased.

28. Results from the Cornell Pesticide Model and the budgeting studies indicate that neither SWCPs nor incorporation of herbicides are highly cost-effective for reducing losses of herbicides in runoff. If implemented only for the purpose of reducing herbicide losses, reductions will generally not exceed 1-2 grams in edge-of-field losses per dollar of farm cost. However, if SWCPs are implemented for the control of soil erosion and/or sediment delivery, reductions in herbicide losses may be an important secondary benefit.

SUMMARY

The major environmental benefit of SWCPs is their usefulness in controlling edge-of-field losses of sediment and total phosphorus from croplands. SWCPs may not be effective at reducing total losses (runoff plus percolation) of nonsorbed pollutants such as nitrate, although runoff losses may be reduced. Total nitrate losses can be controlled by management of nitrogen fertilizer applications. SWCPs implemented for sediment control will also have some additional benefits in terms of reducing losses of pesticides, especially those which are moderately or strongly adsorbed and are persistent. However, in many cases available alternatives are more effective than SWCPs for control of pesticide losses.

SECTION 3

RECOMMENDATIONS

The principal conclusions of the study (Section 2) reflect a recognition that supporting evidence is largely limited to mathematical modelling results and fragmentary field data. The evaluation of economic and water quality implications of soil and water conservation practices (SWCPs) is a relatively new emphasis and more definitive conclusions must await further study. It is recommended that further research be initiated in four major areas.

1. Long-term field studies to measure the effects of SWCPs on either pollutant losses or crop yields have been rare. Hence determinations of cost-effectiveness are generally based on mathematical models which have seen limited validation. Until systematic long-term field studies are complete, it is likely that many uncertainties will remain. However, future field studies must be linked to modelling efforts, since the site-specific nature of nonpoint source pollutants will always preclude experimental study of all relevant variables.

   Technical and economic evaluation of the water quality implications of individual SWCPs and SWCPs in combination should be continued to more clearly identify reductions in pollutant load, the level of management needed for continued effectiveness, and applicability of use. The purpose of such field studies would be two-fold: to provide an indentification of the probable effects of SWCPs on pollutant losses and crop yields and to provide systematic testing of mathematical models which will be utilized as planning tools in most water quality management studies.

2. The linkages between edge-of-field pollutant losses and water quality were only partially addressed in the project, generally by the implicit assumption of a linear relationship. The transport of pollutants from field to stream or aquifer and subsequent movement and behavior of the pollutants in the receiving water are important determinants of water quality. Unfortunately, these phenomena are both imperfectly understood and infrequently measured.

Field and modeling studies to provide information on the fundamental relationships involved in the transport and transformation of pollutants from source to stream and as the pollutants move in the stream will contribute needed understanding on the impact of SWCPs on water quality.

3. In several sections of the report the use of SWCPs for water pollution control was compared with their use for reducing cropland erosion. The point of these comparisons was not to show that erosion and pollution control represent opposing resource management options, but rather to indicate that an erosion control program can be an inefficient way to control water pollution. While this is an important observation, most individuals would agree that the issue is not management of erosion or water quality. It is clearly essential to the nation and the agricultural community that both problems be solved.

   Before this can be done more research is needed to develop systems of practices which can simultaneously and efficiently serve both soil conservation and water quality goals. Such research should be particularly sensitive to the effects of cost-sharing and technical design criteria on the potential of SWCPs for achieving these two goals.

4. Certain components of the study noted that alternative approaches can be used to help meet water quality objectives. These alternatives include integrated pest management and fertilizer and manure management.

   Investigations to identify the effectiveness of these and other nonstructural management approaches to reduce pollutant loads will permit a better understanding of the proper application and value of both these approaches and SWCPs to help meet water quality objectives.

SECTION 4

DEFINITIONS AND QUALITATIVE EVALUATION OF SOIL AND WATER CONSERVATION PRACTICES

Tammo S. Steenhuis and Michael F. Walter

DESCRIPTION OF SOIL AND WATER CONSERVATION PRACTICES

A soil and water conservation practice (SWCP) is an agricultural practice which reduces soil erosion and/or increases water retention. Selected practices discussed in this report are of interest because (1) they are in common use and recommended by the Soil Conservation Service, and (2) data from field studies are available for reference and comparison of results. The included practices were selected from the SCS National Engineering Handbook, Section 2, (1966) and from control of Water Pollution From Cropland, Volume 1 (Stewart et al. 1975). Definitions of other key terms used throughout this report are in Table 4-1.

SWCPs can be divided into two groups. The first, cultural practices, are nonstructural in nature. This group includes no-tillage, conservation tillage contour farming, graded rows, ridge planting, contour listing, contour strip cropping, sod-based rotations, and cover crops. The second group consists of structural practices which usually require off-farm help for construction. The practices in this group include terraces, diversion terraces, grassed waterways, filter strips, and artificial drainage.

Definition of Conventional Tillage

To compare the relative effectiveness of these practices, a base or standard must be defined. For the purposes of this report, conventional tillage is used as the standard and is defined as moldboard plowing, followed by secondary tillage at least three times to smooth and pulverize the soil, and planting and cultivating where appropriate. Operations are assumed to be done up and down the land slope. In general, all operations, including fertilizer application, are assumed to be done in the spring. In some locations, fall plowing and/or fertilizer application may be conventional, and shifting these operations to spring would be classified as a SWCP. Thus, the concept of "conventional tillage" varies with crop grown and location, but in all cases the surface is free from residues for a period of time.

Cultural Practices

No Tillage--

The term no-till refers to a method of planting crops that involves no seedbed preparation other than opening the soil for the purpose of placing

TABLE 4-1. SELECTED KEY DEFINITIONS OF TERMS USED IN THIS REPORT

Adsorption	The adhesion of a substance to the surface of soil particles.
Erosion	Detachment and movement of soil or rock fragments by water, wind, ice or gravity (for purposes of this study, erosion refers only to that caused by water and gravity). Types of water erosion include: Gully Erosion: Advanced stage of erosion which produces large channels which cannot be smoothed over by normal tillage operations. Natural Erosion: Wearing away of the earth's surface by water under natural environmental conditions of climate, vegetation, etc., undisturbed by man. Also called geologic erosion. Rill Erosion: An erosion process in which numerous small but well defined channels only several inches deep are formed. The rills can easily be removed by normal tillage operations. Sheet Erosion: The removal of a thin, relatively uniform layer of soil particles. Also called interrill erosion (Foster and Meyer, 1977). Splash Erosion: The spattering of small soil particles caused by the impact of raindrops on wet soils. The loosened and spattered particles may or may not be subsequently removed by surface runoff.
Sediment	Any solid material, either mineral or organic, that has been eroded, transported and deposited on land or in water.
Sediment Yield	Amount of sediment which moves off a unit area (e.g., edge of field losses).
Sediment Delivery	Sediment that is transported to a specific point, generally a stream.
Sediment Delivery Ratio (SDR)	The ratio between sediment delivery and soil erosion.
Gross Soil Erosion	Soil erosion as defined by the Universal Soil Loss Equation.

the seed at the intended depth, usually accomplished by opening a small slit or by punching holes in the soil. There is generally no cultivation during crop production, and chemicals are used for weed control. In some parts of the country, no-till has a wider meaning. A band 15 to 20 cm is tilled with a rototiller attachment. Unless otherwise specified in this report, the term no-till is exclusively associated with tillage of a very narrow band of up to 5 cm during seeding.

Conservation Tillage--

This term can refer to a wide variety of tillage systems. In the present definition of the Soil Conservation Society of America, it is any tillage system which reduces loss of soil and water compared to unridged clean tillage. Reduction of soil and water loss is accomplished in most cases by protecting the surface with crop residues.

Contour Tillage--

Contour tillage is a traditional SWCP in which field operations, such as plowing, planting, cultivating, and harvesting, are performed following the natural field contour. Because of its ineffectiveness on long or steep slopes, contouring is often combined with other practices such as strip cropping and terracing (Jamison *et al.* 1968).

Graded Rows--

Where poor drainage or the likelihood of extreme storms are a problem, rows on a slight gradient may be used rather than contouring to prevent breaking over of rows. Rows, therefore, should have just sufficient gradient to prevent the accumulation of excess water.

Contour Listing and Ridge Planting--

Neither practice is commonly used at the present time, but they have potential for reducing runoff and soil loss. Contour listing is accomplished by an implement called a lister, which is a double plow, the shares of which throw the soil in opposite directions, leaving the field with a series of alternate ridges and furrows. Row crops may be seeded in the bottoms of the furrows or on the top of the ridge as they are opened up. Ridge planting refers to a practice in which crops are planted on preformed ridges. Crop residues are moved into the furrows between rows to help control runoff and erosion. Ridges are generally on the contour or at a slight gradient.

Contour Strip Cropping--

The practice of growing crops in comparatively narrow strips of land on which the farming operations are performed on the contour is called strip cropping. Strips of grass, close growing crops, or fallow are alternated with those in cultivated crops. Thus, contour strip cropping implies both contouring and crop rotation.

Sod-based Rotations--

Growing different crops in recurring succession is called crop rotation and is one of the oldest methods of maintaining soil fertility, dating back at least to the Middle Ages in Europe. A typical rotation in much of the country has been a few years of a row crop, followed by one year of small grain required as a nurse crop for the establishment of a sod crop. However, recently introduced herbicides and decreased demand for straw have encouraged many farmers to directly seed their hay crops. Both of these alternatives are included in this definition. In this project, the sod crop was assumed to last for at least 3 years and to consist predominately of alfalfa. In practice, other legumes and/or grasses may be used as sod crop while the duration will be determined by the particular economic situation and the type of crop.

Cover Crops--
A cover crop is a close growing crop grown primarily for the purpose of protecting and improving soil between periods of regular crop production. The most common crops are rye and wheat which are planted shortly before or soon after harvest and plowed down or chemically killed in the spring. The typical situation analyzed in this report is a cover crop following corn silage or soybean harvest, as neither crop leaves sufficient residue to protect the soil against erosion.

Structural Practices

Terraces--
Terraces are not the same in each part of the country. This is reflected in the definition of a terrace as an embankment or combination of an embankment and channel constructed across a slope to control erosion by diverting or storing surface runoff instead of permitting it to flow uninterrupted down the slope. Terraces or terrace systems may be classified by their alignment, gradient, outlet, and cross-section. Alignment may be parallel or non-parallel. Gradient may be level, uniformly graded or variable graded. Grade is often incorporated to permit paralleling the terraces. Outlets may be soil infiltration only, vegetated waterways, tile outlets or a combination thereof. Cross-section may be narrow base, broad base, bench, steep back slope, flat channel or channel.

Diversion Terrace--
Diversions differ from terraces in that they consist of individually designed channels across a hillside. They may be used to protect bottom land from hillside runoff, placed above a terrace system, or used in connection with strip cropping to shorten the length of the slope so that the strips can more effectively control erosion. Finally, they also can offer a last-chance possibility for controlling erosion in critical zones: for example, by diverting water away from the beginning of an existing or potential gully.

Grassed Waterways--
Grassed waterways are natural or constructed broad and shallow channels that are planted with erosion resistant grasses. They are used as outlets for terraces, graded rows or contouring.

Filter or Buffer Strips--
Strips of permanent vegetation which are placed above farm ponds, diversion terraces or other natural or man-made drainage channels or at the edge of field to retard flow are called filter strips or buffer strips. They cause deposition of transported material and thereby reduce sediment flow.

Artificial Drainage--
Drainage is the removal of excess surface water or groundwater from land

by means of surface or subsurface drains. Drainage is usually considered as a soil and water conservation practice, although in the true sense of the definition, it is not, as it conserves neither soil nor water.

Extent of Use of SWCPs

Although accurate information on the extent of use of SWCPs is somewhat difficult to obtain, estimates of the area associated with certain practices have been furnished by Dr. R. M. Gray of the U.S. Soil Conservation Service (Gray, 1977, personal communication). These have been summarized for a number of Land Resource Areas (LRAs) shown in Figure 4-1 and are given in Table 4-2. The term "Crop Residue Use" is not included in the definitions of the previous section. Crop residue use implies leaving at least 2 MT/ha of crop residue on the field over the critical erosion period. Conservation tillage means leaving all or part of crop residues on the field throughout the year. Thus, these two practices are mutually exclusive.

Crop-residue use is by far the most widely used practice, accounting for over 60% of land in practices in the LRAs in Table 4-2. Approximately 13% is in conservation tillage and 18% in terraces.

Contour farming and strip-cropping are fairly minor in most regions, but account for 24% and 20%, respectively, of all land in practices in the Northeast (LRAs 101, 140, 145).

There is considerable variation in land in terraces. In LRA 78, 50% of the land is in practices and nearly 40% of the total cropland is in terraces. For several areas (101, 131, 140, 145, 153), terraced areas amount to under 1% of land in practices. In the corn belt areas (102, 108, 111), about 4% of land in practices is terraced.

Although the LRAs included in Table 4-2 are not necessarily representative of the United States east of the Rocky Mountains, they do represent a variety of agricultural areas and can be assumed to typify patterns of SWCP use in that part of the country.

PHYSICAL PRINCIPLES OF SWCPs

As their name implies, soil and water conservation practices are primarily intended to conserve soil and water. In humid areas, conservation of soil is often the primary goal, whereas in dry areas, conservation of water is traditionally given more emphasis. However, since water and soil move together, controlling the movement of one generally affects the movement of the other. This section discusses the physical means by which SWCPs help control soil and water movement from cropland.

The physical effects of soil and water conservation practices are primarily based on control of runoff, rain splash energy and soil structure. These effects are discussed below and summarized in Table 4-3.

Control of Runoff

Reduction of Runoff Velocity--

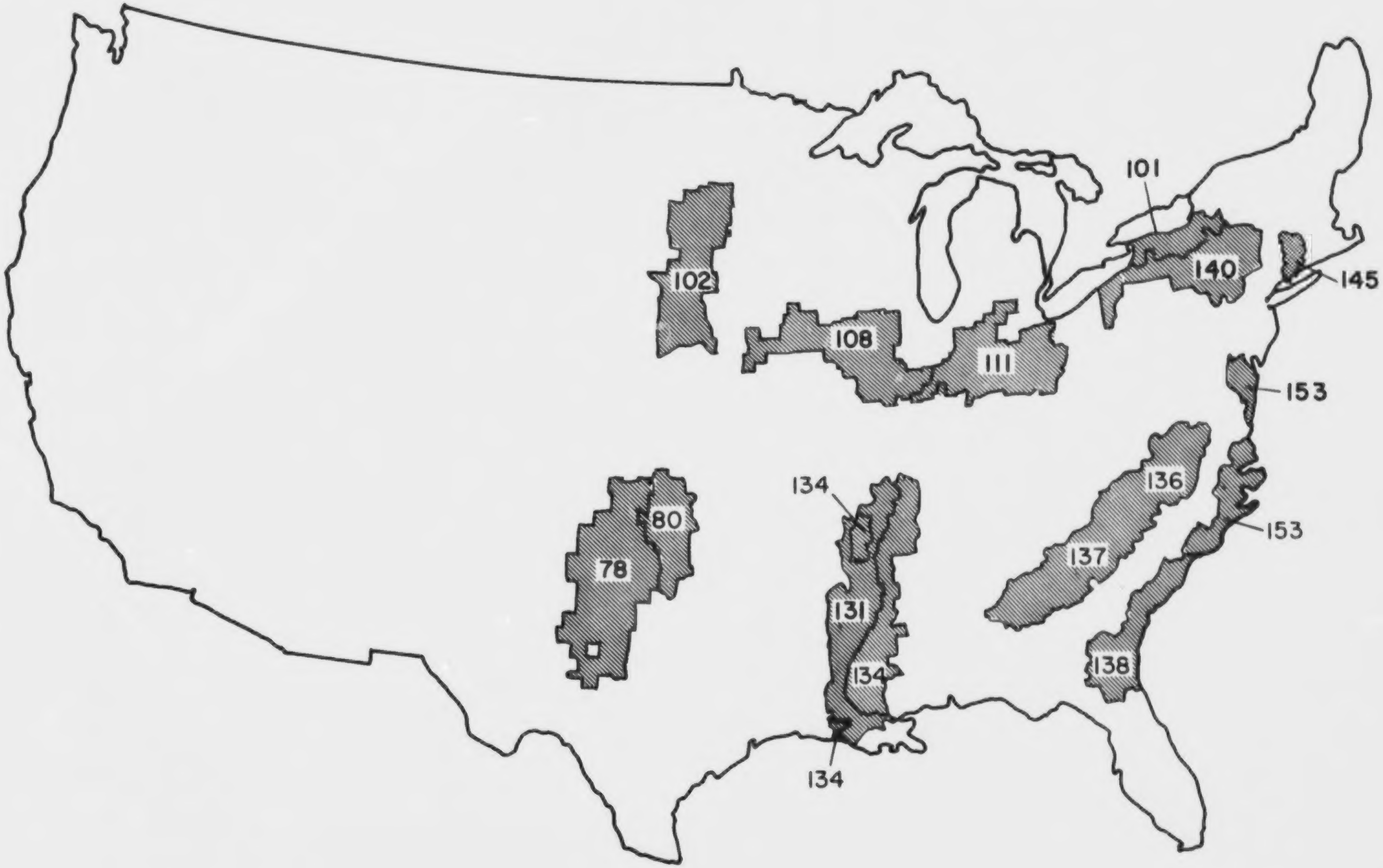
101
140
145
102
108
111
153
136
153
134
80
78
137
131
134
138
134

FIGURE 4-1. LAND RESOURCE AREAS AS REFERRED TO IN TABLE 4-1.

TABLE 4-2. ACREAGE (1000 HECTARES) OF LAND UNDER SOIL AND WATER CONSERVATION PRACTICES AND CONVENTIONAL TILLAGE IN CERTAIN LAND RESOURCE AREAS

SWCP[2]	LRR[1]	H		L	OO		P		T	R						Percent of total area in Practices
	LRA[3]	78	80	101	131	134	136	137	153	140	145	102	108	111	Total	
MT		57	5	43	75	56	65	4	72	26	9*	409	899	856	2567	12
CRU		1085	911	118	2430	1208	262	101	458	62	4	1734	1875	2765	13013	61
CF		202	21	46	15	159	105	17	4	97	1	221	315	28	1231	6
SC		3	3	35	4	1	28	1	*	85	*	78	33	28	300	1
TER		0	33	0	*	0	0	2	0	0	0	0	0	0	35	0
TER, CF		715	123	*	3	17	91	21	1	*	*	162	55	22	1211	6
TER, MT		0	*	0	*	0	0	*	0	0	0	0	0	0	1	0
TER, CRU		0	83	0	*	0	0	1	0	0	0	0	0	0	84	0
SC, CF		7	1	4	*	*	10	1	*	13	0	8	9	*	55	0
SC, MT		*	*	3	*	*	4	*	*	3	0	11	12	5	41	0
SC, CRU		3	4	8	6	4	16	1	*	8	*	52	32	7	142	1
CF, MT		13	1	4	1	4	16	2	*	4	0	42	109	0	205	1
CF, CRU		227	31	11	25	163	61	26	6	8	0	127	278	15	978	5
TER, MT, CF		43	2	*	*	1	15	1	*	0	0*	20	19	3	106	0
TER, CRU, CF		853	274	*	4	19	61	26	1	*	0	40	50	65	1334	6
SC, MT, CF		*	*	*	0	*	1	*	0	1	0	1	3	*	9	0
SC, CRU, CF		1	1	1	0	*	5	1	*	1	0	5	10	*	26	0
Total Practices		3210	1494	274	2564	1633	740	206	544	309	7	2910	3699	3744	21333	100%
Conventional Tillage		922	392	485	1691	888	419	79	454	684	43	2697	2815	3019		
Total		4132	1886	759	4255	2521	1159	285	998	992	50	5607	6514	6763		
% of Total Cropland in Practices		78	79	36	60	65	64	72	55	31	14	52	57	55		

[1] Land resource regions
H = Central Great Plains
L = Great Lakes
OO = Mississippi Delta
P = South Atlantic and Gulf Slope
T = Atlantic and Gulf Coast
R = Northeast
M = Midwest

[2] Soil and Water Conservation Practice
MT = Minimum Tillage
CRU = Crop Residue
CF = Contour Farming
SC = Strip Cropping
TER = Terraces

[3] Land Resource Area
See Figure 4.1 for location

TABLE 4-3. PHYSICAL EFFECTS OF SOIL AND WATER CONSERVATION PRACTICES[a]

	Decrease Runoff Velocity	Increase Surface Storage	Increase Soil Moisture Storage	Decrease Splash Energy	Improve Soil Structure
No-till	+	0	+	+	+,-
Conservation Tillage	+	0	+	+	+
Contouring	0	+	0	0	0
Graded Rows	+	0	0	0	0
Strip Cropping	+	+	+	+	+
Sod-based Rotations	+[b]	0	+	+	+
Cover Crops	+[c]	0	0	+	+
Listing	+[d]	+[d]	-	0	0
Ridge Planting	+	+[d]	-	+	0
Terraces	+	+[e]	0	0	0
Diversions	+	0	-	0	0
Grassed Waterways	+	0	0	+[f]	+[f]
Filter Strips	+[g]	+[g]	+[g]	+[g]	+[g]
Drainage	0	0	+	0	0

[a] + The effect is as indicated
0 There is no effect
- The effect is opposite of the indicated

[b] During the period of sod crop only

[c] During winter and early spring only

[d] Only if plowing on the contour

[e] On level raised bench terraces and tile-outlet terraces with raised outlets only

[f] Only in the waterway

[g] Effective only at the edge of the unit source area

The velocity of moving water will be reduced whenever the total energy available for water movement is decreased. This may be accomplished by forcing the water to move laterally rather than straight down the slope, by reducing the slope of the land through landforming, or by increasing surface roughness which dissipates the water's kinetic energy. Surface roughness is effectively increased by reducing secondary tillage operations, increasing water-stable aggregates on the soil surface or by the use of a mulch cover.

Decreasing runoff velocity helps reduce both surface runoff volume and soil loss. A slower flow rate allows water to remain on the field for a longer period permitting increased infiltration.

Increase in Surface Storage--

Any obstruction in the flow path (e.g., ridges of soil or vegetation) which allows water to pool will increase surface storage. The trapped water is removed from the total surface runoff volume, which results in a decreased runoff velocity and a reduced sediment carrying capacity.

Trial results of measurements of increased surface storage must be evaluated with care, particularly if results are given in percentages. For example, suppose a certain practice increases a field's surface storage capacity from 0.5 to 1 cm of water. For a 2-cm rainstorm, the runoff would drop from 1.5 to 1.0 cm, a 33% decrease. But for a 10-cm storm, the change from 9.5 to 9.0 cm would be less than 6%. Also large rainstorms may wash over and erode the obstructions. Thus, in some cases, for large storms, increased surface storage may not save any more soil or water than if straight row cropping was used (Wischmeier and Smith, 1965).

Increased Conductivity and Moisture Storage--

Some practices increase the soil macropores connecting to the soil surface, which can greatly increase infiltration and conductivity. Soil moisture storage can be increased by either draining or evaporating moisture in the soil profile.

Reduction of the Splash-Energy of Falling Rain--

Rainfall striking bare soil can result in sealing of the soil surface. In many cases a surface crust is a limiting factor for water infiltration (Hillel and Gardner, 1969). Dissipating raindrop energy by use of a plant canopy or a mulch will greatly reduce the surface sealing effect (Duley, 1939) thus increasing infiltration and decreasing runoff volume.

Improvement of Soil Structure--

A change in bulk density, porosity, and percent of water-stable aggregates in soil all affect its erodibility and its capacity for water infiltration. Lower bulk density and higher porosity increase the infiltration rate of soil water. Formation of surface crust is less extensive in soils having a high percentage of water-stable aggregates (Schwab et al. 1966). Practices which reduce tillage operations or increase soil organic matter will generally cause improvements in soil structure.

PHYSICAL PRINCIPLES OF WATER AND POLLUTANT MOVEMENT

Interactions between pollutant movement, climate and SWCPs are so numerous that resource limitations preclude experimental study of all of them (Dean and Mulkey, 1978). Interest in practices for control of agricultural nonpoint sources is recent and few studies of these interactions have been done to date (U.S. Comptroller General, 1977). However, a relatively large body of research is available on the effect of SWCPs on water and sediment movement. This knowledge, together with what is known about pollutant movement, is used in this report to predict the effect of SWCPs on loss of pesticides, nutrients and other chemicals from agricultural land. The physical principles of water and pollutant movement are discussed here to give the reader an insight in the basic processes which are involved, while the movement of soil is discussed in Section 5.

Water Movement

When rain and melt water reach the surface of the ground, they encounter a "filter" that is of great importance in determining the path by which the water reaches the stream channel. The paths taken by the water (Figure 4-2) determine in many cases the type and load of pollutants leaving the field, and hence, the strategies required for decreasing the pollutant load.

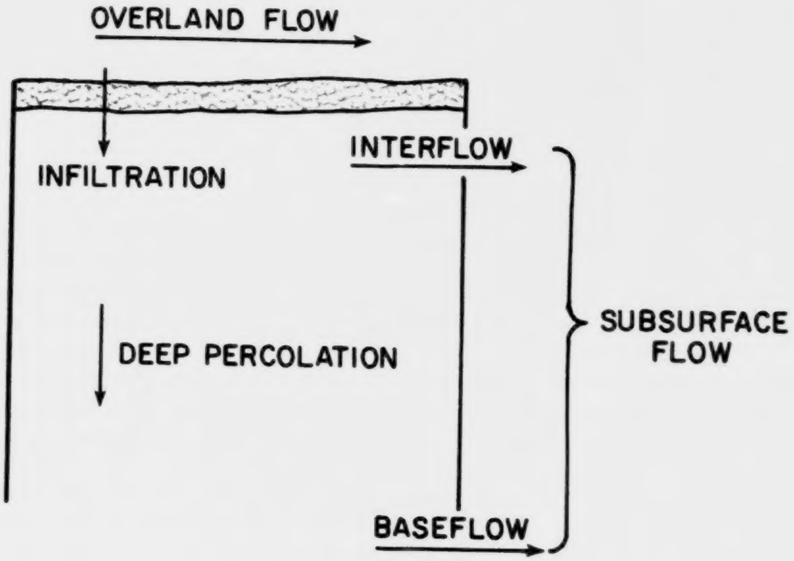

FIGURE 4-2. PATHWAYS BY WHICH WATER MOVES POLLUTANTS FROM CROPLAND.

When the rate of rainfall or snowmelt exceeds the infiltration rate of the soil and the storage capacity of the land-surface depressions, excess water becomes overland flow. Overland runoff will contain dissolved materials such as nutrients or pesticides and carry a sediment load which in its turn may carry adsorbed substances. The quality and quantity of the dissolved chemicals depend on the soil, vegetation, organic residue management, fertilizer and pesticide management, and soil conservation practice.

After water infiltrates the soil, it may be stored there or move out of the field. If the soil is deep and of uniform permeability, the subsurface water percolates to the zone of saturation and then moves with the ground water flow. Because rates of groundwater flow are generally slow and the underground paths long, most of the water following such paths continues to flow between rainstorms. When the groundwater resurfaces in a stream channel it is termed "base flow."

If at some shallow depth in the soil, percolating water encounters an impeding horizon, a portion of the water will be diverted horizontally and will reach the stream by a much shorter route. Because of the shorter route and high permeability of top soil and weathered rocks (relative to unweathered rock), and generally greater gradients of hydraulic potential in these upper sloping horizons, water following this path reaches the edge of the field much more quickly than the groundwater flow described above. This water is classified as subsurface storm flow, or interflow.

In some fields, vertical and horizontal flow may cause the soil to become saturated throughout its depth. When this happens, some of the water moving by the shallow subsurface path emerges from the soil surface and becomes overland flow.

Movement of Pollutants

In this section, only the behavior of pesticides and nutrients are considered. The reader is referred to Section 5 for a discussion of the movement of soil. The physical and chemical properties of pollutants determine their path and mode of travel out of a field. Pollutants can be dissolved in the soil water, be adsorbed to soil particles or be in solid phase.

Water transport of a substance includes movement in its dissolved phase in the surface and subsurface runoff and transport of solid and adsorbed phases in the surface runoff. The volume of runoff water is generally much greater than the quantity of sediment yielded from a watershed. Therefore, even if the concentration of a substance is greater in the sediment than in the water, the quantity of the substances moving off a field dissolved in the runoff might be greater than that adsorbed to the sediment (Mulkey and Falco, 1977; Baker *et al*. 1977, 1978). Both the absolute amounts of a substance carried in the aqueous, adsorbed and solid phases and its relative mobility in the water and sediment are important. Runoff water and the dissolved or aqueous constituents will move farther in a shorter time than sediment and the adsorbed and solid pollutants which are subject to redeposition. The possibility of degradation and transformation is greater during the longer transport time required for sediments.

Grouping Pollutants by Adsorption Properties--

There are many ways to categorize pollutants (solubility, persistence, etc.), but in studying pollutant losses in soil and water, it has been proven useful to classify pollutants, other than sediment, according to their relative concentrations in water and on soil particles as indicated by adsorption-desorption isotherms (White and Beckett, 1964; Ryden et al. 1973). These may be nonlinear and multiple valued for each combination of chemical and adsorbent. However, for the purpose of comparing the adsorption of substances an adsorption partition coefficient, k_s, for a given solution concentration can be calculated as the ratio of amount adsorbed to that in solution, i.e.,

$$k_s = \frac{\text{concentration of substance adsorbed to soil particles (ppm; mg/kg)}}{\text{concentration of substance in solution (ppm; mg/}\ell\text{)}}$$

Typical partition coefficients obtained from published isotherms and runoff studies are given for pesticides and nutrients in Table 4-4 and 4-5, respectively. Some of the members of the pesticide groups exhibit high k_s values, especially with montmorillonite clay. Values with kaolinite clay are lower.

TABLE 4-4. COMPARATIVE GROUPING OF SELECTED PESTICIDES BASED ON ADSORPTION PARTITION COEFFICIENTS OBSERVED IN OVERLAND FLOW WITHIN ONE MONTH AFTER APPLICATION.

Group I $k_s \sim 1000$	Group II $k_s \sim 5$	Group III $k_s \sim 0.5$
paraquat[g] DDT[e] toxaphene[f]	trifluralin[g] atrazine[c,g] cyanazine[b,g] 2-4-D[g] diphenamid[g] alachlor[g] dichlobenil[a] fluometuron[d]	chloramben[f] dicamba[f]

[a] Bailey et al. 1974

[b] Baker et al. 1978

[c] Baker and Johnson, 1977

[d] Davidson et al. 1975

[e] Haan, 1971

[f] Helling, 1971

[g] Smith et al. 1978

TABLE 4-5. COMPARATIVE GROUPING OF SELECTED NUTRIENTS BASED ON ADSORPTION PARTITION COEFFICIENTS

Group I $k_s \sim 1000$	Group II $k_s \sim 5$	Group III $k_s \sim 0.0.5$
Organic Nitrogen	Soluble Inorganic Phosphorus	Nitrate
Ammonium		
Solid Phase Phosphorus		

Many substances are strongly adsorbed to soil and have adsorption partition coefficients of 1000 or more. Another group of potential pollutants, including most currently used pesticides, have adsorption partition coefficients ranging from 2 to 20. Therefore, for the purposes of this study, the following groupings of pollutants are useful.

I. $k_s \sim 1000$, strongly adsorbed and solid phase pollutants (e.g., solid phase organic nitrogen, DDT, paraquat)

II. $k_s \sim 5$, moderately adsorbed pollutants (e.g., most pesticides)

III. $k_s \sim 0\text{-}0.5$, nonadsorbed or soluble pollutants (e.g., nitrate)

Losses of Strongly Adsorbed and Solid Substances (Group I)--

Losses of strongly adsorbed and solid substances in baseflow and interflow are usually small, while losses in overland flow can be high and often are related to the amount of sediment in the runoff and the amount of the substance in the soil. The concentration of the different nutrients vary tremendously for different soils. Brady (1974), for instance, gives a range of 0.02-0.50 mg/liter for nitrogen.

Losses of organic matter in eroded soil can be as high as 1.2 tons/ha (Barrows and Kilmer, 1963). Since organic matter is concentrated in the soil surface and has low density, it is among the first components to be removed (Holt et al. 1970). Thus, sediment contains more organic matter and associated nutrients than the surface soil from which it originates (Massey and Jackson, 1952; Rogers, 1941; Bruce et al. 1975; and McElroy et al. 1976).

Jacobs (1972, quoted by Casler and Jacobs, 1975) and White (1953, quoted by Barrows and Kilmer, 1963) showed that total phosphorus (P) and nitrogen (N) losses are related to the amount of soil carried off the land. When soil loss is high, P and N losses have been positively correlated with soil loss (Duley and Miller, 1923; Timmons et al. 1968), and the concentration of P and N in the sediment and the original soil are usually similar (Massey and Jackson, 1952). But, if the concentration of sediment in the runoff is low, the nutrient concentrations in the sediment can be several times that in the original soil. When soil loss is low, as is often the case with grasslands, there is often no correlation between sediment load and level of nutrient loss.

Paraquat and diquat are known to be strongly adsorbed to soil constituents (Weber and Weed, 1968; Knight and Tomlinson 1967; Burns et al. 1973; Weed and Weber, 1969). Nearly all paraquat and diquat losses are, therefore, associated with sediment (Smith et al. 1978). Nicholson et al. (1966) found that at the outlet of a 400 sq. mile watershed, essentially all DDT was associated with suspended sediment.

The above substances are transported primarily on soil in runoff. Therefore, if a SWCP reduces sediment, it should also influence transport of these substances. SWCPs which deal only with water movement and do not affect sediment movement will not significantly affect losses of these strongly adsorbed substances.

In general, phosphorus is probably best categorized as a strongly adsorbed nutrient. But because it has so many forms, it cannot be as clearly categorized by adsorptive properties as other potential pollutants such as pesticides. There are other complicating issues with regard to phosphorus as well. The origin of phosphorus leaving a field is often difficult to trace. Phosphorus is taken up by the plant and is built into its basic structure. This phosphorus can wash off a field without ever being part of the soil or soil solution.

In the spring, summer and fall, phosphorus losses are mainly associated with the sediment. However, in the winter, at least in the colder climates, phosphorus losses in the dissolved phase tend to be high (Harms et al. 1974). Presumably the cell walls of plant residue freeze and rupture allowing phosphorus to leach out (Martin et al. 1970; White, 1973). Thus, residue management may decrease soil loss but increase the soluble phosphorus losses. McDowell (personal communication, 1978) found that although the soluble phosphorus losses increased due to residue leaching, the total phosphorus load from test fields was lower because of the decrease in soil loss.

Phosphorus has been found in the subsurface flow. Bouldin et al. (1975) estimated that this source accounted for one-third of the inorganic phosphorus load from a New York watershed. The phosphorus concentration in the subsurface flow seems little affected by agricultural practices. Thomas and Crutchfield (1974) found that the amounts of soluble phosphates in the drainage water from a field were essentially the same as published by McHargue and Peter in 1921 despite the fact that fertilizer use had increased from zero up to considerable amounts.

Losses of Moderately Adsorbed Substances (Group II)--

A distinction should be made between the loss of moderately adsorbed substances in overland flow, interflow, and deep percolation. The behavior of substances in overland flow is completely different than that in subsurface flow and the two flow processes are discussed separately.

Although some studies have indicated that losses of moderately adsorbed substances in overland flow are related to both soil loss and runoff water, most evidence shows that they are essentially related to runoff water. The following studies illustrate this point.

In research on the pesticides atrazine and GS13529 applied at rates of 2.2 and 4.5 kg/ha on Hagerstown silty clay loam on a 14 percent slope with corn planted, it was found that over eight times more atrazine was lost in runoff than with sediment at the low rate of application, and over four times at the high rate (Hall *et al*. 1970). Kearney (1972) found much higher losses of 2,4-D, 2,4,5-T and picloram in runoff water than on sediments.

In Iowa, Ritter *et al*. (1974) found that most atrazine was lost in the water fraction. Baker and Johnson (1977) showed that changing tillage practices on a Kenyon soil resulted in a tenfold decrease in soil loss but the atrazine load in the runoff almost doubled even though atrazine application rates were not changed. Runoff water loss remained nearly constant. In Coshocton, Ohio, a strong correlation was found between amount of runoff and total pesticide loss (Triplett *et al*. 1978). Studies in Georgia showed that initial amounts of rainfall abstraction and infiltration were inversely related to pesticide loss. However, there was also a positive correlation of soil loss with total pesticide loss (Bailey *et al*. 1974).

Contrary to movement by surface flow, transport of adsorbed substances by water passing through the soil matrix is relatively slow, and an equilibrium exists between the substances dissolved in the soil water and adsorbed to the soil. Freed and Hague (1974) found that none of the pesticides they tested moved downward more than 50 cm in a loam soil at 25°C under simulated annual rainfall of 150 cm. They looked at a number of pesticides typical of those currently applied to agricultural fields. The worst pesticide pollution of subsurface flow water derives from pesticides that are very weakly adsorbed such as aldicarb (Porter, personal communication, 1977) or that are very slow to degrade.

Studies have indicated that the leaching of strongly and moderately adsorbed substances from the upper horizons does not result in increased concentrations in the groundwater. Simonson (1970) argued that because of the soil formation process in the groundwater zone, losses of nutrients in base flow have been substantial both before and since the appearance of man on earth. Many nutrients dissolved in base flow are, then, of geological origin.

Losses of Non-adsorbed Substances (Group (III)--

Losses of soluble substances in surface runoff are small. Soluble substances are found in very low quantities from small plots where direct overland flow accounts for all surface runoff. With larger areas, subsurface flow, which can transport significant amounts of nitrates and other soluble substances, may emerge as surface flow and contribute to the pollutant load in surface flow (Beasley, 1976; Chichester, 1976). Contributions of nitrates to surface waters by base flow and interflow are at least five times as high as direct overland flow contributions in humid regions (Jackson *et al*. 1973; Campbell, 1976; Booram and Asmussen, 1976). The subsurface flow will contain lower amounts of soluble nutrients when the field is covered with an actively growing crop. The plants will absorb most of the nutrients and percolation will be decreased due to evaporation (Chichester, 1977, Kilmer, 1978 personal communication).

Timing of Substance Movement--

Pesticide losses in surface runoff were found to be highest in the first three storms after surface application (Pionke and Chesters, 1973; Edwards, 1972). For moderately adsorbed substances (atrazine, trifluralin, 2,4-D), the dissolved losses were found to be highest in the first storm after application. The concentration of substances in the sediment decreased with time after application, but much less rapidly than did the concentration of the dissolved substances (Smith *et al*. 1978). Baker and Johnson (1977) and Baldwin *et al*. (1975) concurred with the conclusion that weakly adsorbed pollutants in runoff water decrease rapidly with time. Caro *et al*. (1973) reported a threefold increase in concentration of carbofuran (a weakly adsorbed pesticide) in runoff in the second storm after application over that from the first storm. However, the probable cause of this anomaly was that the pesticide granules were not completely dissolved during the first storm.

A strongly adsorbed pesticide can be found for years after application in the overland flow, depending on its susceptibility to degradation. Harrold and Edwards (1970) and Nicholson *et al*. (1966) among others, showed this to be true for DDT, while studies at Watkinsville confirmed it for paraquat (Smith *et al*. 1978)

EFFECTS OF SWCPs ON POLLUTANT LOSSES

No-Tillage

Since the introduction of the nonresidual herbicides, no-tillage has become more popular each year. Runoff from no-till fields is in most cases significantly lower than from conventionally tilled fields. During five years of monitoring in Ohio, surface runoff volume averaged 1 mm per season for no-till plots while averaging 11 mm per season on conventionally tilled plots (Harrold *et al*. 1970). Thirty-six hours after a 2.6-cm storm, soil moisture to a depth of 105 cm was 2.5 cm greater on no-till plots than on conventional plots, indicating significantly higher water infiltration under no-till (Blevins *et al*. 1977). McDowell (1978, personal communication) found a 90% reduction in overland flow for no-till corn versus conventionally tilled corn. The reduced runoff rate is a direct outcome of the higher infiltration rate (Trichell *et al*. 1968). The higher water transport rate between the surface and subsurface may also cause nitrate ions to infiltrate deeper under no-till than under conventional till (MacMahon and Thomas, 1976; Steenhuis, 1977; Tyler and Thomas, 1977).

No-till soils have a higher conductivity due to both reduced organic matter breakdown and the greater number of pores with ports to the surface. Only channels which are open to the surface can carry tension-free water from the surface to the subsurface (Ehlers, 1973; Dixon and Peterson, 1971). Such ports are formed by earthworms and roots. Increased earthworm activity has been observed in soils under a mulch cover (Graff, 1969; Tetioa *et al*. 1950, Schwerdle, 1969; Ehlers, 1973). If left undisturbed, these channels greatly increased conductivity.

No-till soils are loosened only locally and superficially; yet they have to bear the normal load of traffic in the field. Thus, no-till topsoils tend to be more densely compacted than plowed topsoils (Free, 1970; Harrold et al. 1970; Baeumer et al. 1973). This higher density does not decrease infiltration rate since the increase in the number of surface connected pores outweighs the effect of higher density under no-till. Baeumer and Bakermans (1973) observed that no-till does not result in denser soil if heavy clay is involved.

Shallow soils with an impeding layer at shallow depths form an exception to the rule that no-till decreases runoff compared to conventional tillage (Smith et al. 1974). Runoff starts on these shallow soils when all soil pores are filled with water and there is no more place for water to infiltrate. On these soils, total pore space and not the number of surface connected pores, therefore, determines the runoff volume. Moreover, the effect of increased bulk density under no-till is not offset by the effect of increased earthworm activity; the bulk density is higher and so is the runoff volume.

Numerous studies (Moldenhauer et al. 1971; Harrold et al. 1970; Meyer and Mannering, 1968; Shanholtz and Lillard, 1969; McDowell and Grissinger, 1976) show that no-till is one of the most effective ways to reduce soil loss from a field. Accumulation of organic matter near the surface of the untilled soil causes significantly higher stability of soil aggregates (Blevins et al. 1977). This, as well as the absorption of energy by the impact of falling raindrops and the impedance to water flow by surface trash, reduces runoff volume and velocity. Since lime, organic matter, nutrients and pesticides are applied on the surface, the concentration of nutrients and pesticides in the surface soil is higher in no-till than in conventionally tilled soil because in the first system, these elements are not incorporated into the soil.

Without plowing, surface-applied substances (except soluble substances) are very slowly incorporated into the soil. Sediment, originating from the enriched top layer, has a higher concentration of strongly adsorbed substances such as paraquat, DDT and total N and total P than when the chemicals were incorporated in the soil (McDowell, 1978 personal communication; Smith et al. 1974; Smith et al. 1978). This enrichment is offset by the dramatic reduction in soil loss and runoff such that the overall result is a decrease in total (both aqueous and adsorbed and solid phase) losses of the strongly adsorbed substances.

Losses of plant nutrients (soluble phosphorus and nitrates) in runoff may be higher under no-till than under conventional till due to increased leaching of dead plant residues on the surface of the soil especially during the winter if freezing temperatures break down plant cells, releasing nutrients (Barisas et al. 1978; Johnson et al. 1977; Smith et al. 1974; Timmons et al. 1968). In northern areas where snow melt runoff forms a major portion of the yearly runoff, phosphorus losses in the aqueous phase may outweigh the losses in the particulate phase (Timmons et al. 1970; Harms et al. 1974; Ryden et al. 1973).

The higher infiltration rate and lower runoff velocity both cause reduced runoff and erosion losses from no-till plots. Losses of strongly adsorbed and solid phase pollutants, such as total phosphorus and organic nitrogen will, therefore, be significantly reduced. It is uncertain whether no-till can reduce losses of weakly adsorbed pesticides and plant nutrients (such as dissolved phosphorus) significantly because of the increased plant residue and sometimes higher pesticide use.

Conservation Tillage

Conservation tillage is generally somewhat more adaptable and more suited to colder and wetter soils than no-till (Stewart et al. 1975). However, since part of the crop residue is buried with conservation tillage, the benefit of the remaining mulch on reduction of soil and runoff water losses will be less than with no-till, but greater than with conventional tillage. Conservation tillage will result in a moderate reduction in runoff volume due to increased hydraulic conductivity. Soil loss is reduced by reduction in runoff volume and velocity, and protection from splashing raindrops of the soil by mulch cover.

The reduction in runoff volume for conservation tillage has been shown by many workers. Onstad and Olson (1970) monitored two small watersheds in South Dakota for moisture conservation. A 7.5-cm rainfall produced no surface runoff with conservation tillage, but 2.0 cm of runoff with conventional tillage. Burwell and Larson (1969), and Siemens and Oschwald (1976) showed that various conservation tillage methods increase the time before runoff starts. Laflen et al. (1978) showed that runoff amounts decreased as residue cover increased. Harrold (1960 as quoted by Baver et al. 1972) found that for a three-year average, the relative runoff losses from conventional tillage were twice as high as those from a plow-plant system, a conservation tillage system that deletes secondary tillage. Edwards (1972) found that increasing tillage causes more runoff and less infiltration of water. Studies by Holt et al. (1968) and Falayi and Bouma (1975) showed that with moldboard plowing alone, the initial infiltration and abstraction that takes place before runoff starts are twice as large as when the soil is cultivated. When moldboard plowing is combined with a secondary tillage operation such as disking and harrowing compared to a plow-plant system, initial infiltration and abstraction are greatly decreased. A reduction of runoff volume for conservation tillage is less likely to occur on the shallow stony soils in the Northeast and claypan soils in the Midwest. In New York for example, Free and Bay (1969) found that the runoff for conservation tillage was somewhat higher than for conventional tillage.

For both shallow and deep loess soils, increased surface residue slows runoff water velocity. In Indiana, Meyer and Mannering (1968) found that under steady-state conditions, runoff velocity decreased from 8 meters per minute to 2 meters per minute if the mulch rate was increased from 0 to 2.5 tons per hectare. The reduced runoff velocity together with the protective effect of mulch against raindrop impact decreased soil loss by a factor of 5 or higher depending on the amount of mulch on the surface (Amemiya, 1970; Edwards, 1972; Free and Bay, 1969; Siemens and Oschwald, 1975; and Laflen et al. 1978).

Nutrient losses from conservation tillage should be less than from no-till because some mixing of fertilizer and soil occurs under a conservation tillage system. Still, the dissolved plant nutrient concentration from a field covered with plant material can be higher than for conventional tillage as shown by Johnson *et al*. (1977) and Barisas *et al*. (1978).

Losses of organic phosphorus and organic nitrogen, which are associated with the sediment phase, decrease with increase in cover (Barisas *et al*. 1978; Johnson *et al*. 1977) Conservation tillage increases available phosphorus concentration in runoff and sediment, due to leaching of plant residues but at the same time, decreases runoff volume and sediment loss. Thus, two independent processes determine the direction of the change in loss of available phosphorus amounts of residue, and runoff and soil loss will have an influence on the positive or negative effects conservation tillage on nutrient losses.

The major carrier for strongly adsorbed pesticides is sediment. Thus, tillage practices such as conservation tillage decrease loss of strongly adsorbed pesticides in overland runoff. Baker *et al*. (1978) verified that conservation tillage decreased phosphorus losses as well as losses of strongly adsorbed insecticides.

Moderately adsorbed pesticides are transported primarily with the water, and so reduction in runoff caused by conservation tillage will decrease pesticide losses. However, the reduction in runoff is sometimes offset by an increase in concentration of the pesticide in the runoff as shown by Baker and co-workers (1977, 1978) for atrazine and DCBN. The pesticides presumably leach out of the mulch layer. Thus conservation tillage is most effective in reducing loss of soil, organic matter, and strongly adsorbed pesticides to surface waters. It also has some potential to decrease losses of weakly adsorbed pesticides, but this is dependent on how much pesticide concentration is increased by the amount of pesticides which is leached out of the mulch layer.

Contouring, Ridge Planting, Listing, Graded Rows, Strip Cropping

These practices are grouped together because they are all based on the same principle of creating barriers perpendicular to the natural flow path of water. The primary results are an increase in volume of depression storage and an increase in flow path length. The net results are decreased runoff volume and water velocity. The principal difference between practices will be discussed in relation to their abilities to control water movement and pollutant movement.

Water Movement

Contouring results in very low soil ridges. Weathering over the growing season or intense storms which cause water to wash over the contours can decrease the effectiveness of contouring in reducing runoff volume (Wischmeier and Smith, 1965; Jamison *et al*. 1968). If the slope is too long, contours tend to be ineffective. Contouring in combination with other SWCPs is, therefore, recommended.

On highly erodible soil in southern Mississippi, contoured plots had only 45 percent as much runoff as straight row cropping (Saxton and Spomer, 1968). On the clay pan soil, contouring was effective in reducing runoff in dry years only. Jamison et al.(1968) and Stallings (1945a) summarized and compared older studies on contouring versus straight row in various parts of the country. Reduction in yearly runoff ranged from 20 to 80 percent depending on climate and soil type. The success of contour farming in increasing infiltration of water is greater on permeable soils than on clayey soils as shown by Baver et al. (1972).

Compared with contouring, graded rows have less surface storage. Thus, reduction in runoff volume, especially on the heavier soils should be small. This very small or no reduction in runoff was observed by Moldenhauer et al. (1971).

Contour listing and ridge planting are advocated in the less humid areas of the Midwest. These practices have higher barriers than contouring and are also more resistant to breaking due to the residue left on the land. Ridge planting and contour listing have greater volumes of depression storage and, therefore, greater potential for conserving water than contouring does. Ritter (1971) reported a 61 percent decrease in runoff volume for a ridge watershed versus contouring. Laflen et al.(1978) and Baker et al.(1978) reported that for three hours of simulated rain of 6.4 cm per hour, the percentage of rainfall resulting in runoff for conventional and ridge planting were 70 and 54 respectively. For the same region but larger watersheds, the four year average runoff for conventional tillage was 13% and for ridge planting, 10%, of the natural rainfall.

Strip cropping is based on the principle that checking the downhill flow of water will reduce its capacity both to pick up soil particles and to carry them in suspension (Bennett, 1939). Flowpath and depression storage is not greatly affected by strip cropping and runoff volume will not be affected. This is confirmed by results summarized by Stallings (1945b) from various parts of the United States. Only in New York on a shallow soil did strip cropping reduce runoff, probably due to increased evaporation and consequently, greater effective water-free pore space.

Soil Loss

Contouring, ridge planting, contour listing and strip cropping are all effective in reducing erosion as long as ridges are not broken by runoff (Stewart et al. 1976).

Data by Stallings (1945a) showed that contour farming is much more effective in reducing soil loss than runoff volume.

Jamison et al.(1968) showed that contouring is ineffective in reducing soil loss on long and/or steep slopes unless combined with other SWCPs, because flow concentrates under these conditions. Graded rows have the same effect as contour farming for small storms, but might be more effective for large storms. Large storms cause contour rows to break over. Ridge planting and contour listing are the most effective in keeping the soil on the land. Not only are the ridges less liable to break, but also the velocity of the water is reduced by mulch in the furrows.

Although strip cropping has only minimal effect on runoff volume, it is effective in reducing soil movement. Stalling (1945a) reported soil losses for strip cropping as averaging one quarter of that of similar areas not in strip cropping. The most effective system of strip cropping employs a meadow strip between the tilled strips (Baver et al. 1972).

Pollutant Movement

Losses of strongly adsorbed substances will be reduced by all these practices as long as they retard the soil movement. Contouring would not be a recommended practice on long and steep slopes, as the contours tend to break and become ineffective under these conditions. They might do very well in combination with other practices and on short and moderately sloping land. Ridge planting, contouring and strip cropping are probably the most effective in reducing loss of moderately and strongly adsorbed substances (Baker et al. 1978). Losses of moderately adsorbed substances such as atrazine and DCBN were reduced by ridge planting (Ritter, 1971; Baker et al. 1977, 1978). This was expected because ridge planting, like contour listing reduces runoff volume. Strip cropping and graded rows have only a minor impact on loss of this group of moderately adsorbed pesticides. Contouring might be effective if applied on short slopes. The soluble pollutants are not affected greatly by any of these practices.

Sod-Based Rotations

Rotating row crops with sod crops improves soil structure relative to continuous row-cropping. Plowing the sod under helps increase soil organic matter, while the dense root system of a sod crop increases soil porosity. Rotation studies with cotton, sorghum, a number of vegetables, and clover showed a slight increase in organic matter for those rotations containing clover (Gerard et al. 1962). Other soil properties were not significantly affected. Because of the large amount of sand in the soil, no aggregation occurred.

It is well established that surface runoff from a sod crop will generally be considerably less than runoff from a row crop (Bennett, 1939). This effect is due mainly to the increase in soil porosity. The benefit can continue for the first few years of row crop after the sod crop is plowed under. Experiments at the University of Missouri showed that surface runoff and soil losses in corn following sod were only one-third and one-fifth, respectively, of the amounts from continuous corn (Miller and Krusekopf, 1932). Other tests at Missouri showed continuous corn had twice the surface runoff and several times as much soil loss as corn in a corn-wheat-clover rotation (Bennett, 1939). Plant population densities of corn have increased considerably since the 1930's, so the differences in runoff and soil loss between the two practices has probably decreased, but will still be considerable. On a black-clay soil in Texas, a gradual decrease in soil organic matter was noted when continuous sorghum replaced ten previous seasons of rotations (Adams, 1974). In Georgia on Cecil sandy loam with 7 percent slopes and a three-year rotation of oats-lespedeza, volunteer lespedeza and cotton, the turned-under residues of the lespedeza and oats effectively reduced runoff and erosion during the following cotton year (Hendrickson and Barnett, 1963).

In addition to affecting soil and water movement, rotations have a potential to limit pollution by their effect on soil fertility and pests and diseases in the crops. The increased organic matter and the natural elimination of pests and diseases obviate much of the fertilizer and pesticide usage. Considering all these benefits, rotation appears to be an excellent tool to combat edge of field losses of nutrients and pesticides. One exception might be formed by rotations which include legumes. Recent research has shown that nitrogen losses can be greatly increased in this system (Kilmer, 1978, personal communication).

Cover Crops

This practice has its greatest effect during the non-growing season. It reduces direct runoff and soil loss in the fall, winter, and spring (Uhland and Hendrickson, 1946; Stewart et al. 1975a). Cover crops may decrease leaching of nitrates to the groundwater because of plant uptake. Studies quoted by Stewart et al.(1976) show that non-legume crops are especially effective.

Cover crops might decrease the loss of strongly adsorbed pesticides which usually degrade slowly and are still available in the soil during the winter (Smith et al. 1978). Most of the moderately adsorbed pesticides have short half-lives and by the time the cover crop becomes effective in reducing runoff, pesticide concentration in the water is usually negligible and the effect on total pesticide loss is minimal. Studies by Smith et al. (1978) showed this to be true for atrazine, trifluralin and 2,4-D in Watkinsville, Georgia.

Terraces

Installation of terraces results in decreased velocity of water flow as a consequence of the flatter slope along the terrace as well as the increased flow path length due to lateral movement through water outlets (Spomer et al. 1971; Laflen et al. 1972). Terraces may be divided into three groups on the basis of function: level, graded, and impoundment. Level terraces are used to conserve moisture, whereas the graded terraces are used for orderly disposal of surplus water during times of excess rainfall. Impoundment terraces cause an increase in surface detention storage and, thus, a lower peak runoff (Hanway and Laflen, 1974).

Terraces without a water storage facility are not very effective in controlling runoff volume on the less permeable soils (Stallings, 1945b; Baver et al. 1972). Simulation techniques indicated that for impoundment terraces, 30% of the water may infiltrate due to ponding (Laflen et al. 1972).

Terraces are more effective in reducing erosion that in reducing runoff. Stallings (1945b) reported up to a 100% reduction in soil loss for an experiment in the eastern United States. More recently, Carter et al (1968) found that soil losses were reduced by 50% due to terracing on highly erodable loamy soils of the Southern Mississippi Valley. On poorly managed soils averaging 135 MT/ha annual soil loss, terracing reduced losses to 2.3 MT/ha (Saxton and Spomer, 1968).

Because terraces retain soil on the land, they considerably reduce losses of strongly adsorbed substances such as paraquat and total phosphorus (Smith et al. 1978). However, since terracing does not reduce overland flow greatly, its impact on losses of moderately adsorbed substances is less pronounced, especially on the less permeable soils. Movement of moderately adsorbed pollutants is affected more by the time interval between application and first rainfall than by terracing (Smith et al. 1978).

In the case of impoundment terraces, where both runoff volume and sediment loss are lowered, losses of both strongly and moderately adsorbed pollutants are significantly reduced (Hanway and Laflen, 1974). The reduced runoff volume increased the subsurface flow of water. This subsurface water, having percolated through the soil, had a higher inorganic nitrogen content than the overland runoff (Hanway and Laflen, 1974). Impoundment terraces, therefore, actually increase the loss of soluble inorganic nitrogen to the groundwater below the terraced field.

Grassed Waterways and Diversions

Grassed waterways and diversions collect water and transport it to a point downhill. They are typically designed in such a way that the soil in the waterway does not erode nor does the sediment (from the upland area) settle out.

The construction of a grassed waterway is usually necessitated as a result of the implementation of a SWCP or of the occurrence of a natural feature which concentrates runoff water and necessitates a runoff disposal system. Diversions collect overland flow over a wide area of land when no erosion control measures are employed. Both diversions and waterways concentrate the flow of the water, giving the water less chance to infiltrate. Therefore, these two practices increase runoff volume rather than decrease it.

Waterways prevent formation of gullies in the field (Bennett, 1939). Soil loss from gullies may be high and consist mainly of subsoil. Subsoil is usually low in organic matter and contains little available phosphorus (Ryden et al. 1973) and no pesticides. Little research has been done on the effect of grassed waterways on agricultural land, but recent researchers have looked at the use of grassed waterways for treating runoff water from feed and barnlots (Dickey et al. 1977; Edwards et al. 1970; Bendixen et al. 1969). Excellent results were obtained in reducing the total pollutant load of feed and barnlots to surface waters because most of the water and dissolved pollutants were infiltrated before the end of the waterway. The impression one might obtain from such studies, that pollutant load from agricultural land would also be reduced, is probably incorrect because of the greater volume of runoff from the agricultural fields in the grassed waterway. This greater volume cannot infiltrate. A study by Asmussen et al. (1977) where 2, 4-D was effectively reduced by a grassed waterway should be viewed in the same light. Runoff from their test plots in the grassed waterway was much less than what could be expected under normal farm conditions.

Until more research is done, it seems at present time not justified to conclude that grassed waterways or diversions control any pollutant other than sediment originating from gully erosion.

Buffer Strips

The concept of using a grass buffer strip to remove polluting substances from agricultural runoff is a recent development. Historically, pasture strips, meadow strips, hay or small grain strips were planted as a SWCP. Research on buffer strips for treating agricultural runoff is limited. Therefore, an analogy between buffer strips and strip cropping is made. In this analogy, grassed buffer strips are considered a strip cropping scheme where only the bottom strip is retained. Since the effect of strip cropping on runoff volume was minimal, buffer strips should also have no remarkable effect on runoff. Thus, the effect on moderately adsorbed pollutants is minimal. If formation of channels can be prevented in the buffer zones, sediment and associated pollutants may settle out before they reach the stream. Studies by Doyle et al. (1974, 1977) indicate that buffer strips of 4 to 15 m might reduce the pollutant load on small plots treated with manure. It is uncertain whether these studies can be extrapolated to larger areas. Therefore, more research is required before buffer strips can be recommended as a SWCP for reducing pollutant load to streams.

CONCLUSIONS AND SUMMARY

This section analyzed SWCPs as to their effect on three types of pollutants: strongly adsorbed, moderately adsorbed, and soluble. Table 4-6 is a summary of the effects of SWCPs on pollutant movement. Gneralizations have been made, so that for any particular set of environmental conditions and pollutant characteristics, the effectiveness of a practice might be different from that indicated in the table.

The amounts of strongly adsorbed and solid phase pollutants will be reduced by those practices which reduce sediment loss. Concentrations of moderately adsorbed substances are lower in the water than on the sediment. However, the volume of runoff water removed is much greater than the amount of sediment. Therefore, moderately adsorbed substances are carried mostly with the water. SWCPs which control sediment have only a slight effect on total loss of this group of substances. SWCPs which control total amount of runoff volume should reduce loss of moderately adsorbed substances. Loss of dissolved substances in surface runoff is usually lower than in subsurface flow. Reduced runoff might increase the subsurface flow and thus increase the losses of soluble substances to the interflow and groundwater. If reduced runoff is obtained by increased evaporation and crop uptake, the losses of non-adsorbed nutrients will be decreased.

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TABLE 4-6. EFFECTIVENESS OF SOIL AND WATER CONSERVATION PRACTICES IN CONTROLLING POLLUTANTS[1]

Control of SWCP	Strongly Adsorbed		Moderately Adsorbed	Non-Adsorbed
	Soil, Organic N, Organic P Paraquat	Available Phosphorus		
Effective	no tillage conservation tillage ridge planting sod-based rotations cover crops[2]	sod-based rotations	ridge planting sod-based rotations	
Moderately Effective	graded rows contour listing terraces	graded rows terraces	contour listing no tillage	sod-based rotations
Slightly Effective	contour farming filter strips diversions	no tillage conservation tillage contour farming ridge planting contour listing cover crops diversions filter strips	conservation tillage contour farming terraces	contour farming ridge planting contour listing
Not Effective or Very Slightly Effective	grassed waterway[3] drainage	grassed waterway[3] drainage	graded rows cover crops diversions grassed waterway filter strips drainage	no tillage conservation tillage graded rows cover crops diversions terraces grassed waterway filter strips drainage

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[1] These evaluations have been made for general situations. The site specific effectiveness of a particular practice as it fits into a total farming system might be different than indicated

[2] Not effective for pesticides.

[3] In combination with other practices, grassed waterways are effective.

SECTION 5

THE EFFECTS OF SOIL AND WATER CONSERVATION PRACTICES ON SEDIMENT

Michael F. Walter, Tammo S. Steenhuis and Hanneke P. DeLancey

Soil can be transported by overland runoff and if it reaches a surface water body it can become a water pollutant. Soil is a potential water pollutant because (1) as sediment its mass volume often reduces stream and lake capacities; (2) it can cause high levels of turbidity and settle out on stream bottoms upsetting the stream or lake ecosystem; and (3) sediment can be associated with adsorbed substances which are transported on soil particles. The association of substances with sediment is important because some SWCPs are more effective at reducing the sediment losses than they are at reducing total runoff.

One recent national estimate of natural levels of sediment moving into surface water was 30 percent of the total sediment load (USDA-SCS, 1978). According to this source, approximately 50 percent of all sediment originated from cropland. Other major sources of erosion include that from streambanks, road ditches, construction sites, urban areas, and forests. The erosion from some of these other sources, notably streambanks and road ditches, have very high rates of sediment delivery. Table 5-1 illustrates the relative distribution of sediment from various sources. While in this project, analysis was principally centered on losses of eroded soil from fields, the relationships between cropland erosion and stream sediment levels cannot be ignored. Background levels of sediment and suspended soil particles are an important consideration in evaluating benefits of sediment or suspended soil particle control from cropland because in some watersheds these naturally occurring levels will limit potential water uses.

Consideration of soil loss in runoff is important not only because soil itself is a potential water pollutant but also because some pesticides and nutrients are associated with soil losses. Substances adsorbed to soil move with it in the runoff. Therefore, control of soil loss may directly control losses of adsorbed substances. The interrelationsip between the character of sediment that is associated with adsorbed substances and that controlled

TABLE 5-1. RELATIVE SEDIMENT LOADS FROM VARIOUS SOURCES (AFTER USDA-SCS, 1978)

Geologic	30 percent
Cropland - Sheet and Rill	50 percent
Grassland and Forests	10 percent
Roadside, Streambank, Construction and Mining	10 percent

by various SWCPs can provide a means to estimate the effectiveness of a specific SWCP to control adsorbed substances.

In humid regions, SWCPs have traditionally been designed to control soil erosion. Since some form of soil erosion is the first step in the production of sediment, one could logically expect SWCPs to potentially control soil loss to surface water from areas where the practices are used. One could also extend that reasoning to the control of the movement of soil-associated substances to surface water. But to do so requires the assumption that reduction in gross soil erosion leads directly to similar reductions of sediment and sediment-adsorbed substances reaching the surface waters. This line of reasoning has been the starting point for much of the evaluation of the effects of SWCPs on sediment and substances adsorbed to soil in sediment. However, the factors used to evaluate the effectiveness of SWCPs for erosion control are often different than those needed for analysis of effects on sediment and sediment-associated substances. For example, in addition to the average quantity of soil that moves, information is needed on when it moves, how far it moves, from where it originates and what its character is (e.g., adsorptive properties). These and other factors are considered in evaluating the potential effects of SWCP on sediment yield and sediment associated substances.

Sediment Related Problems

Estimated annual sediment yield in the United States is 1 to 2 billion metric tons (ASCE, 1977; USDA-SCS, 1978). However, sediment load estimates must be made for a specific watershed to be meaningful in terms of water quality. Most watersheds do not have such data although sediment loads to a number of reservoirs are published (Dendy and Champion, 1978).

Reduction in channel capacity and reservoir storage resulting in increased flooding and reduced water supplies is perhaps the most serious detrimental consequence of sediment. A survey in the late 1940's indicated that more than 33 percent of the water supply reservoirs in midwestern states would have to be replaced in less than 50 years due to sedimentation (Beasley, 1972). Many reservoirs are filling at a rate of 5 percent of their capacity per year. Reservoirs with capacities of 123,000 m^3 or less are filling at a rate of 2.7 percent annually. The median storage depletion for all U.S. reservoirs is 1.5 percent per year (McDowell and Grissinger, 1976 after Dendy et al. 1973).

One survey of several hundred reservoirs showed that the rate of sedimentation in small reservoirs is much higher than in large reservoirs (Beasley, 1972). The added cost of providing storage in reservoirs for sedimentation can be significant. Thirty percent or more of the total storage capacity of a reservoir can be designed for sedimentation. Sediment removal from streams, reservoirs and harbors costs $250,000,000 annually by one estimate (ASCE, 1977). Reservoirs which fill with sediment must often be replaced at an even greater cost than initially designed for because of the need to go to secondary sites.

Flood damage which is a direct result of reduced stream capacity due to sediment from agricultural land is difficult to identify. Little quantitative information seems to exist on this topic. Presumably flood damage is increased due to channel clogging by sediment. Estimated losses in the U.S. from sediment and associated flood water damages are one billion dollars annually (Nelson, 1968 after McDowell and Grissinger, 1976). Other sediment related problems include reduced recreational value of water, increased maintenance cost for navigable channels, and drainage problems where the sediment is deposited.

Most of the sediment related problems discussed thus far in this section are a result of sediment accumulation over a relatively long period of time. However, other problems have more short-term impacts. Soil pollutes by muddying the water, inhibiting photosynthesis and increasing oxygen demand (Hartman _et al_. 1977; Mulkey and Falco, 1977). Sparks (1977) lists fish kills as an important problem that results from sedimentation. Silt can clog the gills of adult fish and settle on the spawning beds, destroying the fish eggs. Sparks also reports that sediment can reduce the dissolved oxygen levels of streams which will adversely affect aquatic life. Table 5-2 lists some of the major impacts of suspended soil in freshwater ecosystems. Suspended material, such as clay or organic matter, can increase the cost of obtaining a suitable water supply for some users. All domestic water suppliers and some industrial users of water require that the water be relatively free of soil particles. Industries dealing with food processing must treat sediment-laden water before use.

TABLE 5-2. CLASSIFICATION OF SUSPENDED SOLIDS AND THEIR PROBABLE MAJOR IMPACTS ON FRESHWATER ECOSYSTEMS (MULKEY AND FALCO, 1977 AFTER SORENSEN _ET AL_. 1977)

Sediment Type	Biochemical, Chemical and Physical Effects	Biological Effects
Clay, Silt, Sand	Sedimentation, erosion and physical abrasion, turbidity (light penetration, habitat change)	Respiratory interference, habitat destruction, light limitation
Natural Organic Matter	Sedimentation, dissolved oxygen utilization	Food sources, dissolved oxygen effects

The Environmental Protection Agency has proposed that sediment loads do not exceed 80 mg/l for the protection of aquatic life (EPA, 1973). The above listed problems are presumably from high suspended soil concentrations, which are the result of single events or a series of events over a relatively short period. Where such problems exist, analysis is necessary on a storm event basis.

Soil as a Carrier of Water Pollutants

Those compounds that are carried by soil are usually associated with the colloidal fractions of the sediment. The smaller and lighter the soil particle to which substances are adsorbed, the greater the transportability of adsorbed pollutants per unit soil weight.

Most soil particles have a negative charge at the surface, and will adsorb cations. Adsorption of a particular cation will depend on its concentration and type. The greater the concentration of a particular cation, the more of it will be adsorbed. A cation will be adsorbed preferentially if it has a high valence and/or small size. Organic matter is perhaps even more important as a substance carrier, but the adsorption mechanisms are not necessarily the same as they are for clay.

Nutrients are, besides potentially being adsorbed to soil, an intrinsic part of the soil. Even granite rock is composed of about 0.3 percent P_2O_5 (Black, 1958). For cropping purposes the total quantity of a nutrient in soil is not a particularly useful measurement (Black, 1958). Only the quantity of a nutrient that is available to a crop is generally of interest. Whether nutrients in sediment cause eutrophication response in a manner analogous to field crops is not clear. Early research showed a correlation between total phosphorus loading and increased levels of eutrophication (Vollenweider, 1968). Some researchers who have studied the question more recently argue that the real cause of increased eutrophication is soluble or available nutrients (Lee _et al_. 1978). Very likely the character of the receiving water (e.g., deep versus shallow lake, low versus high pH) will influence the degree of eutrophication resulting from one form of a nutrient to another. But an understanding of the origin of nutrients causing a water pollution problem is critical to an effective solution, because total nutrient loading is often best controlled by reducing sediment losses while available nutrients may not be. Available forms of nutrients are often partly adsorbed to soil particles and are partly in solution. Their control may require runoff controls or better source (e.g., fertilizer) management. Although sediment is typically made up of about 1 kg nitrogen and 0.8 kg of phosphorus per ton (ASCE, 1977), the portions of those nutrients in an available form may be much less, so sediment control may reduce total nutrient losses without really controlling available nutrient forms. Sediment is often higher in clay, silt and organic matter than the soil from which it originated (Massey _et al_. 1953). This phenomenon and the fact that polluting substances have a greater affinity for the smaller particles cause the erosion process to remove the polluting substances preferentially. These two factors: the adsorption behavior of the polluting substances and the sediment characteristics are therefore very important in the evaluation of the effectiveness of SWCPs on nonpoint source pollution control. Adsorption behavior is discussed in Sections 6 and 7 and sediment characteristics are considered later in this section.

Sediments also appear to have a dampening influence on nitrogen and phosphorus concentrations in surface waters (Holt et al. 1970). Glymph (1975) has reported phosphorus adsorption from sewage effluent by sediment. Holt et al. (1970) concluded that adsorption of anions such as phosphate could be an important reaction which occurs in natural colloidal suspensions. These researchers based their conclusion in part on work by Rich (1968) and Chao et al. (1965). Sediments have been shown to remove cesium-137, a major radioactive contaminant of natural waters (Lomenick and Tamura, 1965). Frink (1969) found that potassium was fixed by sediment in lakes. Holt et al. (1970) refer to adsorption of lindane and parathion on the clay-organic complex as "chemical scavenging" of uncharged pollutants in natural waters.

Factors Affecting Erosion and Sediment Transport

Since soil erosion is the first step in producing sediment it is fair to assume that if erosion is controlled sediment will be also. Erosion is very localized so that its control must be field oriented, but sediment can be controlled anywhere between source and sink. In many, if not most instances, erosion control will be the mechanism used to control sediment. But the careful selection of which control practices to use and which eroding soil to control will be necessary for cost-effective sediment control. This selection process requires consideration of the area location relative to sediment polluted water bodies and the type of erosion process producing the pollutant.

The sediment-causing process includes soil detachment, transport, and deposition, and each of these processes has its preference for particular particle characteristics. Soil detachment can occur from either raindrop impact or shear forces of flowing water. Raindrop impact, however, is the primary mechanism responsible for soil detachment at the field surface (Baver et al. 1972; Foster and Meyer, 1977). A number of naturally occurring properties of soil make some more susceptible to detachment. Primary clay particles are relatively less easily detached than sand particles. Particles in the range of 150-250μm (fine sand) are most easily detached by raindrop splash. Eckern and Muckenhirn (1947), Ellison (1947), Bisal (1960), Bubenzer and Jones (1971), Garriels et al. (1974), and Ghadiri and Payne (1977) among others have researched the relationship between rainfall and soil detachment. All concluded that rainfall kinetic energy and drop velocity were important parameters. In general, the higher intensity storms have much higher soil detachment potential than the less intensive storms, even if the storm volume is the same. Free (1960) found that increased rain intensity increased the splash losses of an aggregated soil more than the splash losses from a sandy soil, indicating that a greater percentage of sand is detached at lower intensities. Laws (1941) related drop velocity to drop size and fall height. This relationship has been widely used since soil splash is roughly equal to the product of the drop diameter times the velocity squared (Ghadiri and Payne, 1977). Laws and Parsons (1943) have shown that the mean drop size increases with rainfall intensity which leads directly to the conclusion that the higher the rainfall intensity the greater will be the kinetic energy of that event.

Another feature of splashing raindrops is that they break down the aggregates into smaller aggregates and primary particles, which are more vulnerable to transport than the original aggregates. Detachment by shear forces occurs mainly in areas where water is concentrated (e.g., rills) rather than over a broad area (Foster and Meyer, 1975). Flowing water will detach soil particles if the detachment capacity exceeds the detachment resistance of the soil. As with raindrop splash, fine sand particles are most easily detached. In contrast to splashing raindrops, flowing water does not break down aggregates into smaller units. Hjulström (1935) found that the critical flow velocity for detachment of quartz spheres of clay particle size was about 200 cm/sec, a velocity well above that expected in overland sheet flow. Fine sand size particles on the other hand were detached at less than 20 cm/sec. Foster and Meyer (1977) also describe soil detachment by flowing water in undercutting, sloughing of sidewells, and cleanout of the dumped material.

The second phase of the sediment causing process is the transport of detached soil particles to a water body. Raindrop splash on steep slopes can transport significant amounts of soil downslope, but overland and channel flow are the primary mechanisms for soil particle transport (Young and Wiersma, 1973). Soil particles or aggregates detached by raindrop impact are transported by shallow overland flow to rills. Rainfall significantly increases the transport capacity of shallow overland flow by continuously lifting particles up into the runoff flow (Foster and Meyer, 1977). These particles originate from a very thin layer at the surface and tend to be smaller, due to selective sorting and aggregate breakdown, than the soil *in situ*. Once in defined rills the flow will detach some additional soil but these particles will be similar to the original soil with larger aggregates than those detached by splash (Foster and Meyer, 1977). Once the sediment carrying capacity of the rill flow is exceeded, deposition will begin to occur. Deposition usually occurs due to a change in the carrying capacity of the flow as a result of the decreased flow velocity. Frequently deposition, especially of large particles, occurs long before the runoff reaches streams or lakes (Foster and Meyer, 1977; Meyer and Harmon, 1977).

The transport of detached soil needs to be put into a time frame to be meaningful. Soil particles are often detached and transported for only a short distance in any one storm but after many storms ultimately reach streams. Moore *et al*. (1977) refer to this phenomena as the "play fairs" law, implying that sooner or later sediment deposited in concentrated flow channels will reach streams or lakes. Trimble (1975) has reported that sediment deposited instream channels will move downstream as upstream loading decreases. The idea that sediment deposition increases flow energy, resulting in increased detachment and transport by the runoff, has been widely discussed. McDowell and Grissinger (1976) reported that upland control measures that decrease erosion more than runoff can cause downstream channel instability problems. They discuss some of the theoretical reasons for this phenomenon. For situations where dams have removed sediment load,increased streambank erosion below the dam has occurred (Vanoni, 1977; Graf, 1971).

Sediment is deposited in such a way that larger particles fall out first (Foster and Huggins, 1977). The smallest particles of sediment load, the colloidal component made up of fine clay and organic matter, may stay in suspension for a very long time. So to control the losses of these particles, reduction of their detachment is usually the best mechanism to use.

Rainfall and resulting runoff drive the sediment causing processes. Variations in rainfall and runoff will affect sediment yield. Rainfall can be characterized by its intensity, duration and frequency. SWCPs have no effect on rainfall frequency or duration but these two factors will influence the selection of a SWCP for a particular location. High intensity but relatively short duration thunderstorms, for example, occur more frequently in the midwest than in the northeast of the United States. While SWCPs do not actually change rainfall intensity, they can significantly reduce the energy of the rainfall impact on the soil surface.

Sediment Characteristics

As described above sediment deposited in streams and lakes differs in character from the original soil. Regardless of the source or detachment mechanism, sediment is usually made up of a higher percentage of smaller particles and aggregates than the original soil. This tendency is less when rills and gullies develop in which bigger particles can be attached and transported. As smaller particles have a greater adsorbing capacity, adsorbed substances will be more associated with sediment originating from splash or interrill erosion than with that from rill and especially from gully erosion. This is even more nearly true when chemicals are surface applied; a great part of the sediment which originates from gully erosion comes from deeper layers in the soil profile where the chemicals did not occur. If the chemicals are incorporated into the soil, their concentration is approximate even through the top layer of the soil profile. In this case, only a higher content of small soil particles will increase the amount of adsorbed chemicals in the sediment. To quantify the increased percentage of a specific fraction in the sediment, the term "enrichment ratio" (ER) is used. This term indicates the ratio between the particle fraction in the sediment and that in the original soil. The enrichment ratio for clay (ER_c) is usually greater than one.

Although very little is published on organic matter enrichment (ER_{om}), the transport of organic matter is expected to be much the same as that of clay due to its characteristic low density. It may be that the detachment processes are the same as well. Slater and Carleton (1942) ran two experiments without vegetative cover and found ER_c equalled 1.11 for both and the ER_{om} equalled 1.12 in both cases. These results indicate that clay and organic matter are detached and transported in similar ways.

One of the complications in arriving at ER_{om} is that organic debris can be mixed with the sediment. Values of ER_{om} were 20 to 140 percent greater than ER_c in work reported by Slater and Carleton (1942) for plots with vegetative cover. Residue, leaves, stems, etc., in the sediment will increase the ER_{om}. Relative to adsorbed substances these forms of organic matter probably do not have the same character as clay or soil organic material. Assuming that substances are not adsorbed to organic debris, which may not be so for pesticides, one could reason that practices that rely on vegetative cover or residue control mechanisms, and thus cause low ER_c and ER_{om} (where ER_{om} does not include organic debris), will be most effective, other things being equal, in controlling adsorbed pollutants.

One can generalize that the enrichment ratio for clay or organic matter increases with an increase in soil detachment by raindrop splash relative to that by flow. These enrichment ratios will also increase as transport energy decreases. One shortcoming of the enrichment ratio concept is that it does not indicate the actual amount of clay in the sediment load. A low ER_c for sediment originating from a clayey soil may very well mean a much greater amount of clay in the sediment than a high ER_c from a sandy soil. A high ER_c for a small sediment load may entail less clay than a low ER_c for a big sediment load.

Regional Variations in Erosion

Regions with extended periods of subfreezing temperatures are less susceptible to sediment-producing processes during the winter than warmer areas (Wischmeier, personal communication, 1978). The temporal distribution of intense storms as well as their magnitude varies widely over the humid United States. This is reflected in the extreme regional variation in the rainfall erosivity (R of the USLE) as shown in Figure 5-1. Rainfall erosivity is particularly high in the southern United States. To reach the same level of annual sediment control from two identical fields near, say Ely, Minnesota and Bogalusa, Mississippi, the field near Bogalusa would require five times as intensive controls as that near Ely.

Temporal Variations in Sediment Yield

The combination of erosion index and crop development reflects the seasonal variability in soil loss potential for a particular field. Table 5-3 illustrates the effect of different monthly distributions of the erosion index and crop development for three locations. This table indicates the effect of the distribution of the erosion index and not the quantitative erosivity for the locations. The potential for soil erosion is greatest during the summer months at all three locations but the range between months with high and low erosion potential is most extreme in Iowa.

Although the combination of rainfall patterns and crop management must be used to have an overall view of variations in seasonal erosion potential, some information can be drawn from rainfall patterns alone. Wischmeier and Smith (1965) give rainfall distributions for all the regions covered by the EPA project. Examples for five locations are given in Table 5-4. Although the values do not reflect erosivity due to snow melt some conclusions can be tentatively drawn from this table alone. Control measures for use in Iowa or Minnesota would be designed for maximum effectiveness in the summer. Louisiana,

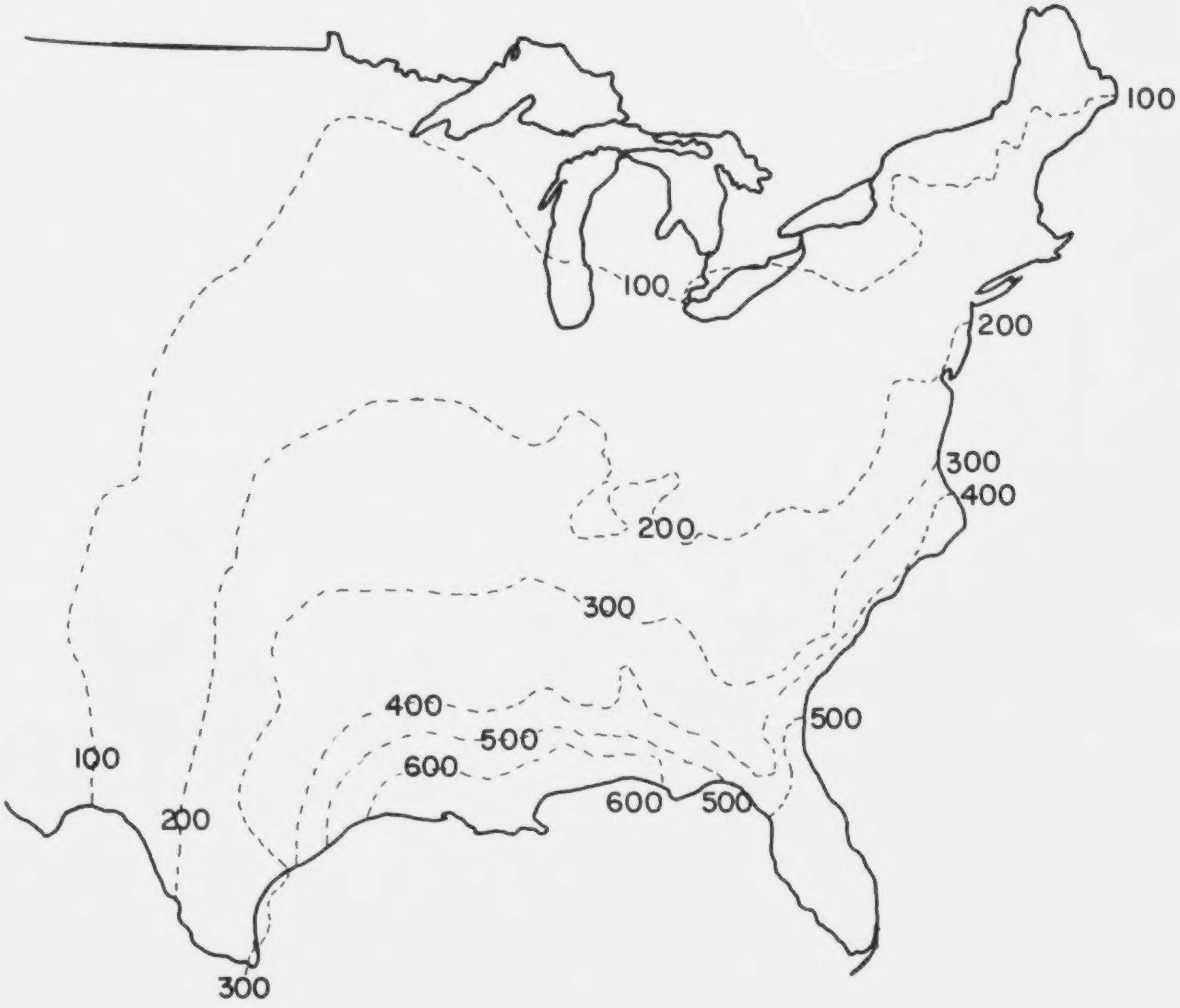
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FIGURE 5-1. AVERAGE ANNUAL VALUES OF THE RAINFALL FACTOR, R (AFTER WISCHMEIER AND SMITH, 1965).

TABLE 5-3. VARIATION OF SOIL LOSS POTENTIAL OVER THE YEAR AS A RESULT OF CROP DEVELOPMENT AND EROSIVITY DISTRIBUTION FOR 3 LOCATIONS WHEN PLANTED WITH CONTINUOUS CORN, SPRING PLOWED AND RESIDUES LEFT

	Watkinsville, GA	Aurora, NY	Ames, IA
J	6.05	.00	.00
F	6.05	.48	.48
M	8.25	.48	.96
A	8.25	.72	1.92
M	17.6	3.48	10.56
J	20.35	7.68	24.64
J	28.33	9.68	12.32
A	9.35	6.88	7.04
S	4.13	2.16	3.52
O	3.03	.8	1.60
N	4.13	.96	0.80
D	1.93	.48	.00

TABLE 5-4. PERCENTAGE OF AVERAGE ANNUAL RAINFALL EROSIVITY INDEX VALUE EACH MONTH (WISCHMEIER, PERSONAL COMMUNICATION, 1978)

City & State	J	F	M	A	M	J	J	A	S	O	N	D
Aurora, New York	0	2	2	3	10	16	22	21	13	5	4	2
Rochester, Minnesota	0	1	1	3	10	23	24	22	11	3	1	1
Ames, Iowa	0	1	2	4	12	29	17	17	11	5	2	0
Lafayette, Louisiana	6	7	8	12	11	11	12	8	6	5	8	6
Watkinsville, Georgia	6	6	8	8	10	12	19	11	7	5	4	4

on the other hand, appears to have fairly constant erosivity index. Potential sediment problems could occur in the winter nearly as easily as any other time so practices should be effective year round.

From the previous discussion characterizing the sediment causing process one would expect that the occurrence of soil loss would vary over the year. This is illustrated by data from Georgia given in Table 5-5 for the ten events resulting in the highest sediment yield from a 1.3-hectare watershed planted in continuous corn from 1973 to 1975. During this three-year period all of the ten highest sediment producing storms occurred during either May, June or July. Spomer _et al_. (1976) found that this pattern also occurred in the upper Midwest. Also consistent with work of Spomer _et al_. is the fact that a few storms produced most of the sediment yield over the three-year period. The ten biggest events produced a total soil loss of 17,500 kg/ha while the 62 smaller storms accounted for only 860 kg/ha soil loss. Since May, June and July account for only 41 percent of the total average annual erosivity index in Georgia (see Table 5-4) one would not expect the ten highest events to be concentrated in these three months. However, this watershed was planted to continuous corn and the effect of crop and residue cover was minimal during May and June. The combination of these two factors made these months highly vulnerable for soil loss. Although the developing canopy cut the erosion potential considerably in July, the erosion potential was still great in this month due to the very high erosion index.

TABLE 5-5. COMPARISON OF PEAK SEDIMENT YIELDING STORMS IN 1973-1975⁺ P-2 WATERSHED, GEORGIA

Storm data	Rainfall (cm)	Runoff (cm)	EI* (foottons/ha)	Sediment Yield (kg/ha)	Concentration (mg/l)
May 28, 1973	10.9	7.4	119.8	8240	11,140
June 11, 1975	7.1	4.0	123.2	4122	10,305
June 6, 1973	3.1	2.2	38.2	1108	5,036
June 27, 1974	10.8	4.2	207.3	966	2,300
May 23, 1973	1.9	0.6	10.8	716	11,930
July 27, 1974	7.2	4.5	288.1	661	1,468
June 13, 1973	2.7	0.7	11.2	530	7,571
July 24, 1975	4.3	2.4	40.6	427	1,779
July 8, 1974	4.8	1.9	37.5	422	2,221
July 13, 1975	2.7	0.8	46.6	320	4,000

*Calculated with USLE method.

⁺Runoff and sediment yield from all storm events in these 3 years are given in Appendix C.

The storm of May 28, 1973, was the largest recorded in the three years at Watkinsville, but only 0.1 cm more rain fell in that storm than in the one on June 27, 1974. However, the sediment yield was almost an order of magnitude greater for the first storm. Total runoff was greater for the first storm

(7.4 cm vs. 4.2 cm). The kinetic energy (EI) value of the second storm was actually greater than the first. The reasons for the May 28th storm producing so much more sediment than the June 27th storm were probably many, but a prime factor was the method of tillage. In 1973 a moldboard plow was used while in 1974 the land was chisel plowed. The 1973 storm apparently resulted in greater soil detachment due to reduced residue cover which caused increased raindrop impact and greater surface sealing which in turn reduced infiltration and increased runoff. An additional factor was that by June 27th some crop canopy had developed which reduced the soil loss potential. This point is perhaps better illustrated by comparing the similar storms of June 6 and July 8, 1973. The sediment yield was over 60 percent less for the second storm for the probable reason that the crop canopy was better developed by the July storm.

Predictive Models

Table 5-5 illustrates that no single parameter can be used to predict sediment yield from a single storm. Wischmeier and Smith (1965) tried to correlate a great many factors with total erosion and finally determined that their proposed method of determining EI values was best. The EI parameter used in the USLE is dependent only on rainfall characteristics. The USLE was not intended to be used for single storm predictions so an anomaly that occurs for some storms is that soil loss is predicted by the USLE even when no surface runoff leaves the watershed. For sediment predictions, particularly if the receiving water is a substantial distance from the eroded area, transport energy estimation is quite important. Therefore, models have been developed that include runoff volume and/or peak flow in an attempt to better predict transport of sediment (See Appendix C).

The three models that were used are defined as the USLE Model, the Williams Model, (Williams, 1975) and the Onstad-Foster Model (Onstad and Foster, 1975). The single storm EI value was used in the USLE Model. The Williams Model uses the same form as the USLE except the R value is replaced by a single event energy term that is a function of total and peak runoffs. The Onstad-Foster Model has an energy term that includes both the USLE single event EI value and total peak runoff. In most situations rainfall data are available but runoff data are seldom readily available so total and peak runoff must be calculated. Total and peak runoffs were calculated for the use in the energy terms as shown in Appendix A.

Total and peak runoff were calculated for the Williams and Onstad-Foster Models. The predictions for total and peak runoff were not very accurate. This, of course, had its influence on the error in the predictions of the soil loss with the Williams and Onstad-Foster models. Therefore, a comparison was made between the predictions of these models with predicted and with observed values for total and peak runoff. The results are presented in Appendix C. All storms for which an energy term could be calculated resulted in predicted sediment yields. Therefore, the USLE and Onstad-Foster models predicted sediment yields for some storms that had no observed or predicted runoff. The energy term in the Williams Model is zero if either total or peak runoff input are zero. Soil loss as predicted by the three estimating techniques is compared with observed soil loss in Appendix C, Table C-1. Except for the largest observed event, the USLE predicted higher soil losses than the Onstad-Foster Model which was consistently higher in its predictions than the Williams model. The USLE over predicts losses from five of the ten largest events listed in Table C-1. It overestimates losses on all of the other 62 events that occurred during 1973, 1974 and 1975.

Sediment Source Controls

The options for source control of soil loss are limited compared to those for many other potential water pollutants. Fertilizer and pesticide applications can be reduced or eliminated but soil is a part of the natural landscape. However, since some soils due to soil character, topography, etc., are potentially more susceptible than others to producing sediment, one control mechanism might be to use these less intensively.

Land use changes to less intensive cropping will by and large reduce sediment losses. But in many situations this alternative is not attractive. There are ways, however, to reduce sediment yield significantly without reducing the intensity of production. Sediment yield can be reduced by selectively increasing cropping intensity on less susceptible fields while decreasing it on those most susceptible to producing sediment. White and Partenheimer (1978) have published results of a study in Pennsylvania that indicated that this selection process might be done on a watershed basis, but most likely it would be more easily implemented on a farm by farm basis. That is, adjustments in field rotations or use within a farm unit while maintaining total production would be easier to implement than adjustments between farms. Ranking of fields as to potential sources of sediment can be done by use of the field's soil erodibility factors (K), slope length factors (LS), practice factors (P) and sediment delivery ratios (SDR). This approach is illustrated for an example farm in New York State in Table 5-6. This information illustrates some of the advantages of analysis on a larger than field unit for more efficient sediment controls.

Even within a field one can to some degree select the origin of sediment by choice of SWCP for sediment control. Those practices that control raindrop impact as mentioned earlier will protect soil at the very surface over the entire field. However, they might have very little influence on rill or gully erosion which tends to cover a more limited area but produces a sediment source deeper in the soil profile. The relative volume of sediment originating from interrill and rill erosion as compared to that which has its source in gullies is difficult, if not impossible to quantify. But correct identification of the problem source is critical in selecting effective practices.

EVALUATION OF SOIL AND WATER CONSERVATION PRACTICES

SWCP Control Mechanisms

Most SWCPs are designed to control either (1) detachment and transport or (2) only transport of soil particles. Depending on the particular pollution problem resulting from sediment or suspended scil particles, choice of a SWCP based on its control mechanism might or might not be important. If sedimentation is reducing reservoir capacity for example, it makes little difference what the character of the sediment is or where it came from. The important element in this case is how much sedimentation occurs. In another situation turbidity might be the critical problem; in which case control of fine particles is the goal and will require a mechanism based on detachment control.

TABLE 5-6. FIELD MANAGEMENT FOR REDUCING SOIL LOSS ON AN EXAMPLE FARM IN NEW YORK STATE

Field Identification	Size (Ha)	System of Crop Rotation Option 1*	Option 2*	Average Annual Sediment Yield*** Option 1 (MT/field)	Option 2
1	23.9	C_s**	C_s	0****	0
2	4.9	C_{g4} H_4	C_g	19.1	35.3
3	3.7	C_{g4} H_4	C_s	5.9	34.4
4	1.6	C_{g4} H_4	C_{s4} H_4	2.6	5.8
5	7.1	C_s	C_{g4} H_4	314.5	54.7
6	1.0	C_{s4} H_4	C_{g4} H_4	17.0	7.7
7	1.6	C_g	C_{g4} H_4	22.6	12.3
8	3.6	C_{g4} H_4	C_{s4} H_4	10.1	22.3
9	2.6	C_{g4} H_4	C_{s4} H_4	7.0	15.3
			TOTALS	398.8	187.8

*Option 1 and 2 have the same acreage in C_g, C_s and H

**C_s - Corn Silage Continuous

C_{g4} - Corn Grain 4 Years

H_4 - Hay 4 Years

C_g - Corn Grain Continuous

C_{s4} - Corn Silage 4 Years

***Based on USLE and SDR (Walter _et al._, 1978)

****Slope essentially equal to zero so erosion predicted to be zero

Table 5-7 lists some general pollutant forms and the potential for their control by detachment or transport mechanisms. Choice of control mechanism may depend partially on climate, soil character, topography and especially field crop management.

Mechanisms for soil detachment control are often more effective for clay soils because clay is more easily controlled at its source than in transport. For rainfall splash control a broad area is typically covered by vegetative

TABLE 5-7. POTENTIAL EFFECTIVENESS OF DETACHMENT AND TRANSPORT CONTROLS ON SEVERAL CATEGORIES OF SOURCES OR CAUSES OF POLLUTION

Source or Cause of Pollution	Control Mechanism	
	Rainfall Transport	Flow Transport
Clay	High	Low
Sand	Medium	High
Aggregates	High	Medium
Adsorbed Substances	High	Low
Organic Matter	High	Low
Suspended Loads	High	Low
Bed Loads	Low	High
Sediment Yield	High	High

canopy or crop residue. The requirement for broad coverage is sometimes in conflict with the farm operations. Sod, for example, will prevent soil detachment almost completely but many farmers have only limited use for sod. These SWCPs are effective in reducing soil detachment by intercepting raindrops and dissipating their energy. The amount of reduction in raindrop impact energy depends on canopy height and density when vegetative cover is used to reduce detachment. Figure 5-2 indicates that, for SWCP that include crop canopies, the erosivity as given by the USLE would be reduced 80 percent or more when the ground is completely covered. Schottman (1978) found that crop canopies tend to break up raindrops so that, for intense storms, effectiveness is likely even greater than shown in Figure 5-2. At least for most agricultural canopies, control of soil detachment is approximately directly related to the percent of ground area which is covered by the canopy. If the canopy is formed by sod forming crops, the soil erodibility is also reduced, as a result of a sod-residual effect (Wischmeier, personal communication, 1978).

Crop residues might be considered an extreme situation for crop canopies, as the canopy is at the soil surface. Comparable percent ground cover residues or "mulches" are more effective at reducing erosion than crop canopies as can be seen by comparing Figures 5-2 and 5-3. Residue reduces both soil transport, and detachment by flow and raindrop impact.

To this point the discussion of crop canopies and residue has centered on their effectiveness in reducing raindrop impact. However, ground cover has other positive effects on the sediment-causing processes as well. Ground cover decreases surface sealing and increases surface roughness which increases infiltration and tends to slow overland runoff and reduce detachment by flow.

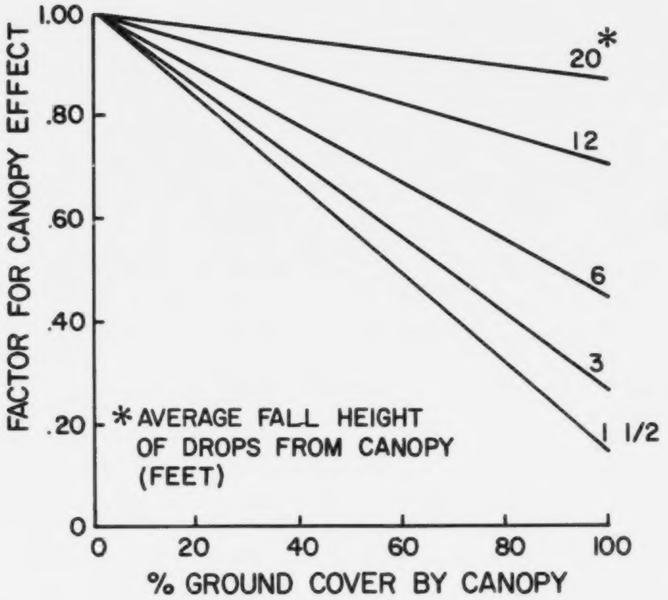

FIGURE 5-2. EFFECT OF CROP CANOPY ON EFFECTIVE EI (ADAPTED FROM STEWART ET AL. 1975).

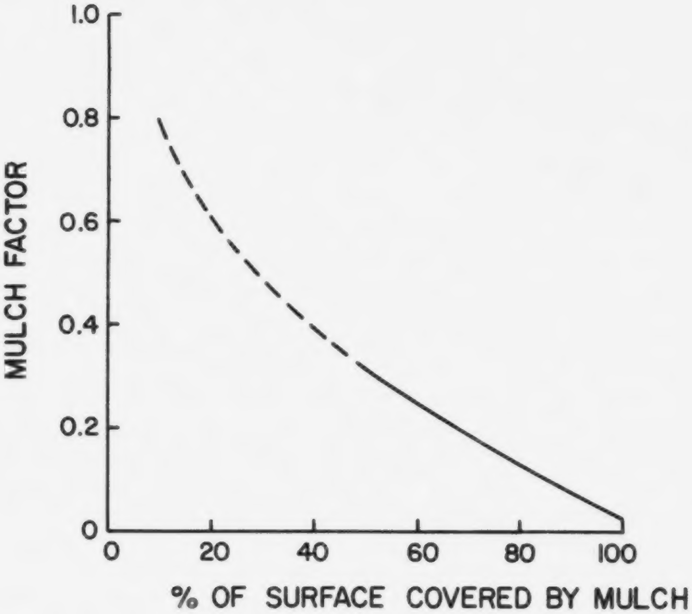

FIGURE 5-3. EFFECT OF PLANT-RESIDUE MULCH ON SOIL LOSS (ADAPTED FROM STEWART ET AL. 1975).

Runoff is the primary vehicle for transporting soil particles once they are detached although under special situations raindrop splash will also transport particles downslope. Transport controls include techniques of reducing both runoff velocity and volume. In general, but not always, when either is reduced so is the other.

The mechanisms used to reduce runoff velocity or volume can be either macro or micro in scale. For example, graded terraces are constructed at the base of a field strip. The primary purpose of graded terraces is to collect and concentrate runoff so that it can be carried off on a gentle slope in a designed channel. The mechanism to reduce slope is to increase flow path on a macro scale. Graded rows have essentially the same purpose (i.e., increase flow path) but act on a much smaller broader scale. Both of these SWCPs will also increase temporary surface storage in the terrace channels or crop rows which can lead to deposition, increased infiltration and many other interrelated processes. Although few data are available on comparison of terraces vs. contour tillage or graded rows, one might expect some differences due to their areal distribution. Terraces do nothing or very little to prevent interrill erosion but do reduce flow detachment and influence transport of soil off a field. Contours do not reduce soil detachment by raindrop impact either but they do cause deposition to occur much sooner in the process. The whole point of controls that operate on soil particle transport is to cause sedimentation to occur on the field. Presumably the sooner sedimentation occurs the better.

SWCP Effectiveness

The best information where available, to judge effectiveness of practices at reducing soil erosion is that used to quantify C and P factors on the USLE. Table 5-8 lists some common cropping systems together with C values, the factor used to reflect the effect of crop cover on annual soil erosion. Wischmeier and Smith (1965) and Beasley (1972) list 128 different cropping systems and Beasley gives a method for calculating a C value for any system. Published values of the P factor are given in Table 5-9. Most SWCPs affect both soil detachment and transport but each generally is primarily effective at one or the other control mechanisms. The categorization of SWCP by control mechanism is similar, though not the same as to division of cultural and structural practices given in Section 4.

Soil Detachment Control Practices

On an average annual basis one can easily see the effectiveness of vegetative cover from Table 5-8. By simply planting corn, even continuous corn that is fall plowed with residue removed, the expected average soil erosion is reduced 40 percent as compared to fallow conditions. Inclusion of a small grain into a two-year rotation cuts the potential average erosion in more than half again. A rotation of corn-wheat-meadow reduces soil erosion by an order of magnitude when compared to fallow conditions. Continuous meadow essentially eliminates soil losses. The effectiveness of some SWCPs are functions of

TABLE 5-8. SELECTED C VALUES FOR COMMON PRACTICES WITH CONVENTIONAL TILLAGE (AFTER STEWART ET AL. 1975)

Rotation	Range of Values for Crop Factor (C factor of USLE)
Corn - Residue Removed, Fall Plow	0.54 - 0.62
Corn - Residue Removed, Spring Plow	0.50 - 0.59
Corn - Residue Left, Fall Plow	0.42 - 0.52
Corn - Residue Left, Spring Plow	0.38 - 0.48
C-O*	0.17 - 0.25
C-C-C-W-M - Residue Left	0.17 - 0.23
C-C-W-M-M - Residue Left	0.09 - 0.14
C-C-W-M - Residue Left	0.07 - 0.11
C-W-M - Residue Left	0.06 - 0.10
Cotton	0.34 - 0.49
Grass	0.00 - 0.01
Alfalfa	0.02
Soybeans	0.48 - 0.54
W-M	0.05
W-Fallow	0.38
Fallow	1.00

C-Corn, O-Oats, W-Wheat and M-Meadow

how much residue is left on the surface. This is true for practices such as chisel or no-till plowing. Table 5-10 lists a few practices and the typical percent ground covered with residue when they are used. Baker et al. (1978) concluded that percent of soil covered by crop residue explained between 78 and 89 percent of the variance in erosion among six different tillage systems.

Some judgments as to effectiveness of SWCPs to prevent soil detachment are possible by relating the time-dependent rainfall erosivity and crop cover with critical periods of pollution by suspended solids. Two primary considerations in evaluating suspended solids are 1) the time during the year when sediment concentrations are highest, and 2) the time period when water quality problems related to suspended solids occur. The expected time of highest erosion potential due to soil splash detachment can be predicted by combinations of the time dependent erosion index distribution and the crop cover. Sediment produced by splash erosion is more likely to stay in suspension than that from other types of erosion. Critical pollution periods are a function of the nature of the problems. For example, settleable solids

TABLE 5-9. VALUES OF SUPPORT PRACTICE FACTOR, P (AFTER STEWART ET AL. 1975 AND WISCHMEIER AND SMITH, 1965)

Practice	Land Slope (percent) 1-2	2-7	7-12	12-18	18-24
1. Contouring[b]	0.6	0.5	0.6	0.8	0.9
2. Contour-Terraces	$0.6\sqrt{n^*}$	$0.5\sqrt{n}$	$0.6\sqrt{n}$	$0.8\sqrt{n}$	$0.9\sqrt{n}$
3. Contour Listing or Ridge Planting or Contour Strip Cropping[a]	0.3	0.25	0.3	0.4	0.45
[a]Recommended Strip Widths (in meters)	31	31-27	27-23	23-18	18-15
[b]Slope-Length Limits (in meters)	120	120-90	90-25	25-20	20-18

n* is the number of equal length intervals the field is divided into.

TABLE 5-10. THE EFFECT OF RESIDUE LEFT ON SURFACE BY SELECTED TILLAGE PRACTICES ON RUNOFF AND EROSION (AFTER BAKER ET AL. 1978)

Tillage Practices	% Residue Remaining on Surface	Runoff * (mm)	Erosion * (MT/ha)
Conventional	5	65	15.4
Buffalo Till	16	39	6.2
Chisel	19	58	10.2
Disk	30	49	5.4
Ridge-Plant	35	37	2.0
Coulter (no-till)	56	39	0.9

*Averaged result of three simulated storm events on three soils.

destroy fish eggs during the incubation period but turbidity can foul drinking water at any time. Crop and residue cover when used as part of a sediment control system differ from some SWCPs in that their effectiveness is very seasonally dependent. This variability is reflected in the C value of the USLE.

Wischmeier (personal communication, 1978) has divided crop effectiveness for erosion control (primarily soil detachment, see Stewart et al. 1975) into six stages. Crop cover factors can be determined for each of the following six periods:

Period F : Inversion plowing to secondary tillage.

Period SB : Secondary tillage for seedbed preparation until the crop has developed 10% canopy cover.

Period 1 : End of SB period until crop has developed a 50% canopy cover.

Period 2 : End of period 1 until canopy cover reaches 75%.

Period 3 : End of period 2 until crop harvest.

Period 4 : Harvest to plowing or new seeding.

The actual periods of the year covered by each of these crop stages depend on the location and particular cultural practices. The monthly crop cover values for the USLE when combined with the monthly percent of average erosion index result in products that reflect seasonal soil detachment potential. Table 5-11 illustrates these seasonal detachment potentials for four crop management systems for corn: spring and fall plowed with residue left, spring plowed with residue removed, and minimum tillage in Watkinsville, Georgia. Crops are generally planted in a rotation so that the variability is even more pronounced.

To determine the average annual erosion potential a procedure similar to that shown in Table 5-11 is used except that "crop-stage periods" are used instead of months. Cover factors such as those given in Table 5-8 are general values. The actual values used by SCS are specific to a location and include rainfall patterns. If a table for C values for monthly or crop stage periods can be found for a location of interest they can be used directly to evaluate time effectiveness of some practices. However, these values are not readily accessible so they need to be calculated as shown in Wischmeier and Smith (1965) and Beasley (1972).

The annual average cover factor for the year of corn grown using systems B (spring plow, residue removed), C (fall plow, residue left), D (spring plow, residue left), and E (minimum tillage) of Table 5-11 is 0.59, 0.49, 0.40, and 0.24, respectively. But the important point relative to single event controls is that certain practices work better than others at any given time. That is, the cover factor is not uniformly reduced or increased from one

TABLE 5-11. RELATIVE EFFECTIVENESS OF SELECTED PRACTICES AT VARIOUS TIMES OF THE YEAR IN WATKINSVILLE, GEORGIA, FOR CONTINUOUS CORN

	J	F	M	A	M	J	J	A	S	O	N	D
A. Fraction of Annual Erosion Index	.06	.06	.08	.08	.10	.12	.19	.11	.07	.05	.04	.04
B. Residue Removed-Spring Plow*	.69[a]	.69	.69	.69	.76	.71	.64	.33	.27	.27	.69	.69
A X B	.041	.041	.055	.055	.076	.085	.122	.036	.019	.014	.028	.028
C. Residue Left-Fall Plow†	.57[a]	.57	.57	.57	.68	.65	.54	.31	.21	.21	.39	.57
A X C	.034	.034	.046	.046	.068	.078	.103	.034	.015	.011	.016	.023
D. Residue Left-Spring Plow*	.37[a]	.37	.37	.37	.64	.62	.54	.31	.21	.21	.37	.37
A X D	.022	.022	.030	.030	.064	.074	.103	.034	.015	.011	.015	.015
E. Minimum TillΔ	.30[a]	.30	.30	.30	.21	.21	.20	.17	.11	.21	.30	.30
A X E	.018	.018	.024	.024	.021	.025	.038	.019	.008	.011	.012	.012

*Assumes: plowed May 1, planted May 15, harvested October 31.
†Assumes: plowed November 15, planted May 15, harvested October 31.
ΔAssumes: planted May 15.
[a] Soil loss ratio as defined in Wischmeier and Smith, 1965.

practice to another for all months. In the example of Table 5-11, the minimum tillage system is most effective at precisely the period when erosion rates are highest from June to September. The practice of leaving residue in the field is about as effective as minimum tillage in the winter months but not during the critical summer period.

SWCPs that are relatively ineffective in late spring and early summer are likely to be not very effective for sediment control, at least not for continuous corn cropping. Crop canopies and residue cover are very seasonally dependent, while most structural practices such as terraces are not.

Whenever crop rotations are used,certain years can be expected to have greater potential for soil loss than others. For example, the probability of soil loss increases each year after sod if corn is planted in a sod-based rotation. This is illustrated in Figure 5-4. A sediment control program needs to recognize these critical years within a rotation and a management system should be developed that intensifies control practices during critical years where possible. For example, tillage practices which leave more residue might be used during the third and fourth years of corn following sod.

Tables 5-12 and 5-13 list the probability,as determined by a 25-year simulation with Williams' model, of daily soil loss in tons per hectare for similar fields in Aurora, New York, and Watkinsville, Georgia, respectively. The lower probabilities of soil losses in New York as compared to Georgia are primarily due to climatic conditions. Because of the nature of the factors in the model no soil losses are predicted from December through March in New York even though some small losses could be expected. However, soil losses in New York are small enough in the winter so that winter controls (e.g., spring versus fall plowing) would not be as effective as in Georgia. No daily soil losses greater than 5 tons per hectare were predicted for the 25 years simulated in New York but 20 such events could have been expected in Georgia.

Soil Transport Control Practices

Many of the practices listed in Section 4 function by reducing transport capacity of runoff. The support-practice factor (P) of the USLE reflects effectiveness of SWCPs to reduce soil transport. Of the six factors included in the USLE, the P factor is quite often considered to most reflect the effect of a SWCP. Actually, P values are available for only a few SWCPs and they are based on "limited field data" in comparison to the other USLE factors (Stewart et al. 1975). Table 5-9 lists published P values for various SWCPs.

The effectiveness of transport control of a SWCP is dependent on the magnitude of the runoff event. Unfortunately, since every situation is a little different it is difficult to quantify how effective a particular practice is under situations of a large storm event. However, this is an important point because, as was shown in Table 5-5, relatively few storms produced most soil movement from the 1.3-hectare fields in Watkinsville, Georgia, and similar results were found for the Black Creek Study in Indiana (Lake and Morrison, 1978).

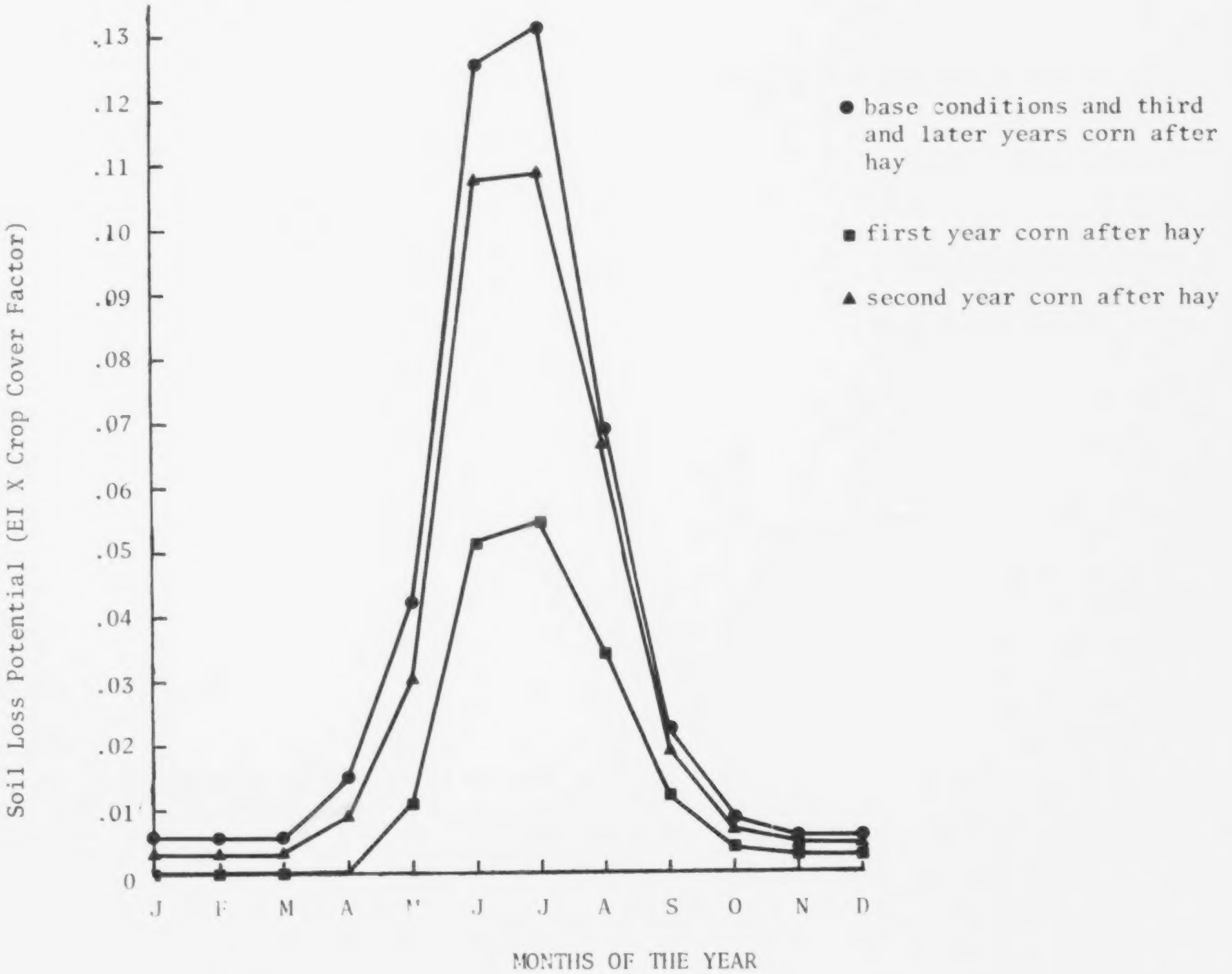

FIGURE 5-4. RELATIVE POTENTIAL FOR SOIL LOSS OVER THE YEAR FOR CONTINUOUS CORN AND FOR CORN AFTER HAY IN NEW YORK.

TABLE 5-12. PERCENT PROBABILITY PER MONTH OF THE OCCURRENCE OF A SOIL LOSS PRODUCING STORM EVENT IN AURORA, NEW YORK, WHEN CORN IS SPRING PLOWED AS WITH WILLIAMS' MODEL IN A 25-YEAR SIMULATION

Month	Zero	Soil Loss (Tons/Hectare)													
		0.01 to 0.49	0.50 to 0.99	1.00 to 1.99	2.00 to 2.99	3.00 to 3.99	4.00 to 4.99	5.00 to 5.99	6.00 to 6.99	7.00 to 7.99	8.00 to 8.99	9.00 to 9.99	10.00 to 10.99	11.00 to 11.99	12.00 to More
1	100.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2	100.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
3	100.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
4	99.8	-	-	-	-	-	-	-	-	-	-	-	-	-	-
5	100.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
6	99.3	0.1	-	0.1	0.2	-	-	-	0.1	-	-	-	-	-	-
7	99.8	-	-	0.1	-	-	-	-	-	-	-	-	-	-	-
8	99.6	0.3	-	-	-	-	-	-	-	-	-	-	-	-	-
9	99.7	0.1	-	-	0.1	-	-	-	-	-	-	-	-	-	-
10	99.6	0.1	0.1	0.1	-	-	-	-	-	-	-	-	-	-	-
11	100.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
12	100.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-

TABLE 5-13. PERCENT PROBABILITY PER MONTH OF THE OCCURRENCE OF A SOIL LOSS PRODUCING STORM EVENT IN WATKINSVILLE, GEORGIA, WHEN CORN IS SPRING PLOWED AS PREDICTED WITH WILLIAMS' MODEL IN A 25-YEAR SIMULATION

Month	Zero	Soil Loss (Tons/Hectare) 0.01 to 0.49	0.50 to 0.99	1.00 to 1.99	2.00 to 2.99	3.00 to 3.99	4.00 to 4.99	5.00 to 5.99	6.00 to 6.99	7.00 to 7.99	8.00 to 8.99	9.00 to 9.99	10.00 to 10.99	11.00 to 11.99	12.00 to More
1	92.1	5.2	2.0	0.1	0.2	0.1	-	-	-	-	-	-	-	-	-
2	92.7	5.3	1.1	0.5	0.1	-	-	-	-	-	-	-	-	-	-
3	91.8	3.8	1.9	1.6	0.2	0.2	0.1	-	-	-	-	-	-	-	-
4	94.4	2.4	1.8	1.2	-	-	-	-	0.1	-	-	-	-	-	-
5	92.9	1.4	1.4	1.5	1.1	0.3	0.2	0.2	0.1	0.1	0.2	-	-	-	0.1
6	96.2	1.0	1.0	0.9	0.1	0.1	0.1	-	-	-	-	-	-	-	0.2
7	97.2	0.7	0.7	0.2	0.5	0.1	0.1	-	-	-	-	0.1	-	-	-
8	97.9	0.9	0.6	0.3	0.1	-	-	-	-	-	-	-	-	-	-
9	97.7	1.4	0.2	0.4	0.1	-	-	-	-	-	-	-	-	-	-
10	98.7	0.7	-	0.5	-	-	-	-	-	-	-	-	-	-	-
11	94.9	3.0	1.2	0.4	0.4	-	-	-	-	-	-	-	-	-	-
12	93.8	3.6	1.8	0.6	0.1	-	-	-	-	-	-	-	-	-	-

Soil transport control by SWCP is reduced in effectiveness as slope length and storm size increase. The practices are designed to control or intercept relatively shallow overland flow. In the case of contouring, contour listing or ridge planting flow will concentrate if the slope length is too long or the storm event large and will eventually break over the ridges, potentially creating an even worse problem. Recommended spacing for terrace or strip crop intervals (Table 5-9) is based on preventing concentrated flow that could produce deep rills or gullies. As the magnitude of a runoff event increases, the effectiveness of a SWCP to control soil transport decreases. For example, the relative effects of increased surface roughness or infiltration on runoff reduction are less for large storms.

Grass buffer strips are placed between streams and upland agricultural land for the purpose of removing pollutants from overland flow. The principal control mechanism of grassed buffer strips is to reduce runoff velocity so that particles settle out. Infiltration might also be increased so that runoff will be reduced. Some of the factors which might influence buffer strip effectiveness are width of strip relative to runoff volume, density of grass cover, detention time of flow in buffer strip and perhaps most importantly, uniformity of flow over the buffer strip.

The effectiveness of terraces is not time dependent as most crop canopy or residue inclusive SWCPs are. However, the effectiveness of ridge planting, contour strip cropping, and contouring may vary with time. Wischmeier (personal communication, 1978) shows ridge planting most effective in crop stage periods SB and 1 (seedbed and establishment, respectively). These periods typically span May, June and July, and as illustrated in Figure 5-4, these are at least for continuous corn, potentially very vulnerable periods for soil loss.

Following the categorization of Section 4, the cultural and structural practices are compared in Table 5-14 relative to their effectiveness of control of each mechanism. The table should be used with care because the effectiveness of a practice is site specific.

Selected SWCP Effect on Soil Loss

Four different management systems were analyzed for a 25-year period in Georgia for continuous corn. The field physical features were those of watershed P2 in Watkinsville, Georgia. In all cases the field was tilled on the contour. The four systems analyzed were 1) spring moldboard plow, 2) fall moldboard plow, 3) spring chisel plow, and 4) no-till. Assumptions were that fall moldboard plowing was done November 15, spring moldboard and chisel plow on May 1, emergence date was May 15, the crop was harvested on November 1st and had a yield of 4 metric tons/ha while 2900 kg residue was left per ha.

The predicted probabilities of a storm event capable of producing runoff are given in Table 5-15. Probabilities of surface runoff are highest in winter and lowest in late summer and fall. Table 5-16 lists the probabilities of a soil loss producing storm event on a monthly basis as predicted with three different models (the USLE, the Onstad-Foster and the Williams Models). In

TABLE 5-14. QUALITATIVE EFFECTIVENESS OF SELECTED SWCP TO CONTROL VARIOUS STAGES OF SEDIMENTATION PROCESS

Practice Type	Raindrop Detachment	Runoff Detachment	Runoff Transport	Seasonal Effectiveness* F	SB	1	2	3	4
Cultural									
No Tillage	H**	M	M		H	H	H	M	M
Cons. Tillage	M	L	M		M	M	M	M	L
Contour Farming	N	L-M	L-M[1]						
Graded Rows	N	L-M	L-M[1]						
Contour Strip Cropping	M	M-H	M-H[2]						
Sod Based Rotations	L-H	L-H	L-H[3]						
Cover Crops	L-H	L	L		M	L	L	M	H[4]
Contour Listing	N	M[7]	M						
Ridge Planting	M	H[7]	M		H	H	M	M	M
Structural									
Terraces	N	H[8]	H[5]						
Diversions	N	H[8]	L						
Grassed Waterways	N	H[8]	L						
Filter Strips	N	L	L-M[6]						
Artificial Drainage	N	N	L						

*Crop stage periods (Wischmeier, personal communication, 1978).
**H - High, M - Medium, L - Low, N - No Significance or none.

1. Effectiveness depends on slope and slope length, see Table 5-9.
2. See Table 5-9.
3. H for years when in sod, L for other years.
4. Level of effectiveness depends on degree of canopy development, see Figure 5-2.
5. Effectiveness depends on the reduction in transport energy which will depend on outlet design.
6. Effectiveness depends on size and character of filter strip and uniformity of overland runoff.
7. Controls detachment in rills.
8. Controls detachment in gulleys.
9. Greatest effectiveness during years with sod.

TABLE 5-15. PERCENT PROBABILITY OF DAILY RUNOFF FROM THE P2 WATERSHED IN WATKINSVILLE, GEORGIA OVER A 25YEAR PERIOD FOR BASE CONDITIONS. PREDICTIONS WITH SCS METHOD

Month	Zero	0.01 to 0.24	0.25 to 0.49	0.50 to 0.99	1.00 to 1.49	1.50 to 1.99	2.00 to 2.49	2.50 to 2.99	3.00 to 3.49	3.50 to 3.99	4.00 to 4.49	4.50 to 4.99	5.00 to 5.49	5.50 to 5.99	6.00 to More
1	86.4	6.4	2.7	1.4	1.8	0.5	0.2	0.2	-	-	-	-	-	-	0.1
2	84.1	8.9	2.8	2.1	0.7	0.5	0.2	-	0.1	-	0.2	-	-	-	-
3	87.2	6.1	1.2	2.0	0.7	0.6	0.2	0.3	0.1	0.2	0.1	0.2	-	0.2	0.1
4	92.4	3.7	0.6	1.2	0.8	0.6	0.1	-	0.2	-	0.1	-	-	-	-
5	92.6	3.4	1.1	1.4	0.3	0.3	0.1	0.2	-	-	-	-	-	-	0.1
6	96.2	2.4	0.5	0.2	0.2	-	-	-	-	-	-	-	-	-	0.2
7	97.2	1.8	0.2	0.5	-	-	-	-	-	-	-	-	0.1	-	-
8	97.8	1.6	0.2	0.2	-	-	-	-	-	-	-	-	-	-	-
9	97.3	1.8	0.1	0.2	0.1	0.1	0.1	-	-	-	-	-	-	-	-
10	98.4	1.0	-	0.1	0.2	-	-	0.1	-	-	-	-	-	-	-
11	91.7	4.1	2.0	0.9	0.1	0.4	0.2	0.2	0.1	-	-	-	-	-	-
12	88.3	6.5	1.1	1.6	0.7	0.2	0.6	0.2	-	0.1	0.1	-	-	-	-

TABLE 5-16. PERCENT PROBABILITY OF A STORM EVENT PRODUCING SEDIMENT (GREATER THAN 0.1 T/HA) FOR FOUR DIFFERENT PRACTICES AS PREDICTED WITH 3 MODELS ON THE P2 WATERSHED IN WATKINSVILLE, GEORGIA

	USLE MODIFIED				ONSTAD-FOSTER				WILLIAMS			
Month	SP	FP	CP	NOT*	SP	FP	CP	NOT	SP	FP	CP	NOT
1	10.5	10.9	10.9	9.7	8.3	8.4	8.0	7.9	7.9	7.9	7.7	7.5
2	10.4	11.2	10.1	9.2	8.1	8.3	8.0	7.3	7.3	7.7	6.8	6.1
3	12.4	12.4	12.3	11.9	10.2	10.2	9.9	9.2	8.7	8.8	7.9	7.8
4	9.1	9.1	9.1	8.8	7.8	8.4	7.5	7.1	5.6	5.6	5.4	5.1
5	11.3	11.3	11.1	9.6	11.3	11.1	9.7	7.8	7.1	7.0	6.8	6.1
6	9.5	9.5	9.2	7.2	9.2	9.2	7.5	5.2	3.8	3.8	3.8	3.0
7	10.1	10.1	10.1	9.0	10.1	10.1	9.2	5.0	2.8	2.8	2.5	2.2
8	8.0	8.0	7.9	7.4	7.9	7.9	8.1	4.7	2.1	2.1	1.9	1.7
9	7.0	7.1	6.3	4.8	5.6	5.9	4.7	3.2	2.3	2.6	2.2	1.8
10	5.6	5.7	5.1	3.5	4.3	4.7	3.0	2.1	1.3	1.3	1.2	1.1
11	7.9	8.2	7.9	7.5	6.4	6.6	6.4	6.3	5.1	5.1	5.0	5.0
12	9.9	9.9	9.7	9.1	6.6	7.0	6.5	6.5	6.2	6.2	6.2	5.9

*SP- Springboard Moldboard Plow, FP-Fall Moldboard Plow, CP-Chisel Plow, NOT-No Till all cases the Williams Model predicts less soil loss than the other two. It also indicates more variation in the probability of soil loss over the year. This greater variation in soil loss probabilities is not surprising since the Williams Model is more dependent on runoff than either of the other two. The monthly runoff probabilities are quite variable.

Probabilities of a soil loss producing storm event from the spring moldboard plowed system are lowest in September, October and November. The Williams Model indicates that it is quite low in June and July as well but the other two do not. None of the models predict much difference in fall vs. spring plowing relative to probability of a soil loss producing storm event. The USLE and Onstad-Foster Models suggest a very slight increase in soil loss probability from about December to May; the Williams Model does not. On the other hand all of the models predict reduced probability of soil loss relative to moldboard plowing when fields are chisel plowed. They predict even greater reductions when no till is used.

The probability of an event causing more than one ton of soil loss per hectare is given in Table 5-17. This table indicates when severe soil loss is likely to occur. For these larger events the Williams Model seems to agree with the others. Apparently the small storm events are included by the USLE and the Onstad-Foster Models but not by the Williams Model. The point was made earlier that the USLE and Onstad-Foster Models can predict soil losses for runoff events even if no runoff is predicted.

Under the spring plow system the most severe soil loss can be expected in May, June and July according to the USLE Model. This was precisely the conclusion reached by analyzing three years of sediment data from the P2 watershed (see Table 5-5). The Onstad-Foster and Williams Models support this conclusion also but they indicate a somewhat more uniform distribution throughout the year. April is also predicted to be a somewhat troublesome month.

When comparing spring versus fall moldboard plowing for severe soil loss events (i.e., greater than 1 t/ha) all the models predict increased vulnerability from December through April with the fall plowing. In January, for example, the three models predict from 3 to 5 percent chance of a one ton per hectare soil loss event with spring plowing as compared to 9 to 18 percent chance with fall plowing. Over the whole year the probability of a severe soil loss increases about 25 percent when the field is plowed in the fall instead of in the spring.

Chisel plowing greatly reduces the potential for severe soil loss events throughout the year. This is particularly true from April to September when the probability of severe events is 50 to 60 percent less than when they are spring moldboard plowed.

No till plowing reduces soil loss 70 to 80 percent over spring moldboard plowing. It is most effective in May through September but substantially reduces soil loss from severe events in all months.

TABLE 5-17. PERCENT PROBABILITY OF A STORM EVENT RESULTING IN SOIL LOSS GREATER THAN ONE TON PER HECTARE FOR 4 DIFFERENT PRACTICES AS PREDICTED WITH THREE MODELS ON THE P2 WATERSHED IN WATKINSVILLE, GEORGIA

	USLE MODIFIED				ONSTAD-FOSTER				WILLIAMS			
Month	SP	FP	CP	NOT*	SP	FP	CP	NOT	SP	FP	CP	NOT
1	0.3	0.9	0.3	0.3	0.5	1.4	0.4	0.3	0.4	1.8	0.4	0.3
2	0.3	1.2	0.1	0.1	0.9	1.4	0.3	0.2	0.6	1.4	0.3	0.1
3	2.0	3.6	1.6	1.0	2.2	3.4	1.7	1.5	2.1	3.5	1.7	1.5
4	1.6	2.6	0.9	0.7	1.4	2.6	1.0	0.6	1.3	2.9	1.1	0.2
5	3.9	3.9	1.8	0.9	3.0	2.8	1.8	0.7	3.8	3.5	1.7	1.2
6	3.9	4.0	0.9	0.2	1.9	2.1	0.4	0.2	1.4	1.5	0.4	0.2
7	4.4	4.6	1.4	0.4	2.1	2.0	0.7	0.1	0.9	1.0	0.3	0.1
8	2.2	2.2	0.9	0.3	1.1	1.1	0.3	0.1	0.4	0.6	0.1	0.0
9	1.1	1.3	0.4	0.3	0.4	0.5	0.3	0.1	0.5	0.5	0.1	0.1
10	0.8	0.8	0.5	0.0	0.4	0.4	0.3	0.0	0.5	0.5	0.1	0.0
11	0.6	0.8	0.3	0.4	0.6	0.9	0.6	0.5	0.8	1.1	0.6	0.5
12	0.6	1.5	0.3	0.3	1.0	1.9	0.6	0.5	0.8	1.9	0.6	0.3

*SP-Spring Moldboard Plow, FP-Fall Moldboard Plow, CP-Chisel Plow, NOT-No Till

This analysis suggests that small soil losses are going to occur under all four of the systems. Even when one goes from spring to fall moldboard plowing to no till, only modest reduction can be expected in the probability of soil loss producing storm events. But the probability of severe soil loss events is greatly reduced by practices such as chisel plowing or no till cultivation. In other words, the number of soil loss producing storm events stays approximately the same, but the amount of soil which is lost is greatly reduced. As compared to fall plowing, spring moldboard plowing reduces soil loss potential in the winter. Chisel plowing is most effective in the spring and summer, but is better than either spring or fall moldboard plowing in all months. No till reduces the potential for serious soil losses substantially over the other three systems from March through November and is comparable to chisel plowing in the winter months.

SUMMARY AND CONCLUSIONS

The sediment causing processes include soil detachment and transport. The bulk of on-field soil detachment is a result of rainfall impact. Both primary particles and aggregates can be detached. Sand and silt are more easily detached than clay. Aggregates tend to be broken down by continuous raindrop impact. Soil detachment by rainfall occurs in a very thin layer at the soil surface. Soil detachment by runoff can occur if shear forces of flow are great enough or if flow undercuts the soil. Detachment by flow is localized in areas where runoff concentrates and generally comes from deeper in the soil profile than that detached by rainfall. Aggregates detached by runoff tend to be larger than those detached by rainfall.

Soil is transported by runoff. Clay and organic matter are more easily transported than coarser particles such as sand. The energy of runoff is reflected in its velocity and volume.

Sediment characteristically has smaller and lighter particles than the original soil. Sediment deposition in streams and lakes is not the only form of water pollution from soil; soil can also be in suspension or move as a bed load. The suspended load which can cause turbidity, reduced photosynthesis, etc., will be made up predominately of clay and organic matter. Suspended load is relatively variable over the year and is typically the result of a single or series of intense storm events. If the suspended load originated as soil detached by rainfall, it is more likely to have adsorbed chemicals than if it is made up of soil detached by runoff, especially if the chemicals were surface applied. Bed load typically consists of coarse material. The bed load does not fluctuate as much as the suspended load and is a minor element in terms of soil adsorbed substances.

All practices designed to control erosion will potentially reduce sediment losses. Their effectiveness at erosion control is reflected in the cover and supporting practice factors of the USLE. The effectiveness of SWCPs at sediment control will depend on field location and to a lesser degree on the sediment producing storm characteristics. Soil and water conservation practices that control soil detachment as a result of rainfall impact (e.g., crop canopies and residue cover) will not be as sensitive to storm intensity as SWCPs based on transport control (e.g., contour tillage and terraces). The effectiveness of some SWCPs that control transport decrease as the magnitude of runoff increases (e.g., contour tillage and lister planting).

Raindrop detachment controls are typically crop canopies or residue covers. These controls are more effective at preventing movement of adsorbed substances than transport control SWCP. Detachment of soil by runoff is controlled by intercepting or slowing overland runoff by increased surface roughness before a flow becomes concentrated. Transport control of soil is accomplished by reducing runoff velocity or volume. Transport controls rely on soil settling out of runoff so that they are more effective at removing coarse than fine particles.

The potential for a storm event to produce sediment varies regionally and over the year for a given region. A few very intense storms can account for most of the sediment yield from a watershed.

Certain practices are more effective during one time of the year than another. This is particularly true of vegetative canopies and residue covers. Seasonal practice effectiveness should be matched with seasonal sediment production potential. A comparison of four tillage systems - fall moldboard plowing, spring moldboard plowing, chisel plowing and no-till in Watskinsville, Georgia, showed that none of the practices greatly reduced the probability of the number of soil loss producing storms, but the probability of severe events (i.e., greater than 1 ton/ha sediment yield) was significantly different. As compared to spring plowing, fall plowing resulted in about 25 percent more severe events mostly occurring in the winter; chisel plowing reduced the probability by 40 to 50 percent; and no-till cultivation reduced the probability of severe soil loss by 70 to 80 percent.

SECTION 6

EFFECTS OF SOIL AND WATER CONSERVATION PRACTICES ON EDGE-OF-FIELD NUTRIENT LOSSES

Douglas A. Haith

This section presents the results of modelling studies which were used to quantify the effects of certain SWCPs on edge-of-field losses of nutrients. The rationale for the use of mathematical models is two-fold. Although as noted in Section 4 qualitative conclusions regarding the effects of SWCPs on nutrient losses are possible, relatively few field data are available which would permit a quantitative assessment. An objective of these modelling studies was to provide this quantification for certain practices and locations. A second objective was to develop a general methodology for evaluating SWCPs which would be applicable to a variety of crops, locations and practices.

The importance of such a methodology should not be underestimated. Nutrient losses from croplands are site-specific, depending on weather, soils, nutrient management and cropping practices. No SWCP can have a uniform nutrient control effectiveness (e.g., 10% reduction in nitrogen losses) for all locations and crops. In the absence of very extensive field studies, models are essentially the only means of generating site-specific information.

There is a variety of information which is of interest in a quantitative assessment of SWCPs. Edge-of-field losses of nitrogen (N) and phosphorus (P) dissolved in runoff and in percolation waters from a field are of concern as are the solid-phase losses of these nutrients associated with eroded soil. This information can be reported in several ways, such as losses for specific time periods or storm events and average monthly and yearly losses. The most straightforward evaluation of SWCPs is a comparison of their effectiveness in controlling the mean or average annual nutrient loss from croplands. These losses are reported in this section, based on either ten or twenty-five year simulations.

An evaluation of SWCPs in terms of their ability to control mean annual nutrient losses is not completely adequate, however. Since nonpoint source pollution is influenced by weather conditions, nutrient losses in runoff and percolation can be highly variable. Thus losses may be very high in certain years and quite low in other years. This variation can be presented as frequency distributions of annual nutrient losses. At least conceptually, such distributions can be used to evaluate the risks (probabilities) of large nutrient losses associated with various SWCPs. For example, two alternative SWCPs may both reduce mean annual dissolved P loss in runoff by approximately the same amount, but one of the practices may result in a much larger variance of annual losses than the other. It then could be inferred that the risk of P losses significantly larger than the mean is greater with the high variance SWCP. In many instances this would indicate that the alternative SWCP which has a comparable mean, but smaller variance is preferable. Accordingly SWCPs are evaluated in this section both in terms of mean annual losses and frequency distributions of annual losses.

This section also discusses the simulation models used in the estimation of nutrient losses, model calibration or validation and the long-term (10-25 yr) simulation of SWCPs for four locations (central New York, central Iowa, northeast Georgia, and central Michigan). These locations were chosen primarily to reflect the effects of climatic variability. While results should not be extrapolated directly to other locations, they are indicative of the different levels of effectiveness that may be expected from SWCPs. The SWCPs analyzed were conservation (minimum) tillage, contouring, terracing, sod-based rotations, and no-tillage.

Although this report is primarily an evaluation of SWCPs, it must be recognized that other agricultural management practices may be suitable for control of cropland nutrient losses. In particular, nutrient losses may be closely related to the quantities and timing of fertilizer applications. While a systematic evaluation of fertilizer management was not attempted in the project, the simulation models were used to estimate the effects of changes in N fertilizer applications on losses of the nutrient in runoff and percolation. This work is summarized in Appendix D.

SIMULATION MODELS

Two general simulation models were used in the analysis of nutrient losses. The first of these is the Agricultural Runoff Management (ARM) model which was developed by Hydrocomp, Inc. for the U.S. Environmental Protection Agency (Donigian and Crawford, 1976; Donigian et al. 1977). The second model was developed at Cornell for a related research project (Tubbs and Haith, 1977) and will be referred to as CNS (Cornell Nutrient Simulation) for convenience. The general features of the two models are outlined in this section, while more detailed descriptions are given in appendices A (CNS) and F (ARM).

The two simulation models differ significantly in structure and data needs. The CNS model is based on the U.S. Soil Conservation Service (SCS) curve number runoff equation and the universal soil loss equation. The impacts of SWCPs on CNS model parameters can be obtained from readily available sources, and model calibration is unnecessary. The model provides a general computational tool which incorporates the most commonly used runoff and soil loss estimating procedures and can be applied to arbitrary crops, soils and locations in the absence of field water quality data. In contrast the ARM model attempts a less empirical approach to the estimation of runoff and soil loss and is in principle more general than either the curve number or universal soil loss equations. This generality is achieved through the use of calibration parameters and the model can only be applied to field situations which have been monitored for runoff. Since ARM has been calibrated on only two experimental fields (one each in Georgia and Michigan), the model is presently of limited value in the analysis of regional differences in effectiveness of SWCPs.

As might be expected from the above discussion, the evaluations of SWCPs reported in this section are based primarily on the CNS model. However, certain practices, most notably no-tillage, cannot be studied using CNS, largely because runoff parameters (curve numbers) for this condition are unknown. The ARM model provided a means of studying no-tillage and comparing its effects with other SWCPs such as contouring and terracing. Also, since the CNS model is based on the SCS curve number equation, it predicts only total or direct runoff and does not separate a runoff hydrograph into surface and subsurface flow components (overland flow and interflow). Conversely, the ARM model estimates both runoff forms and was used to study the effects of certain SWCPs on nutrient losses in both surface and subsurface runoff.

Agricultural Runoff Management (ARM) Model

In the current application, the ARM model computes water, soil and nutrient losses from a field in surface runoff and interflow at 15-min intervals during a runoff event. During other times, soil nutrient levels are updated every 6 hrs. Vertical movement and transformations of water and nutrients in the soil profile are simulated in surface, upper, lower and groundwater zones or layers. The depths of the surface and upper soil zones are specified by model input parameters and are generally 0.3-0.8 cm and 8-15 cm, respectively. The upper zone depth corresponds to the depth of soil-incorporated minerals which are available for loss in overland flow (surface runoff). Nitrogen and phosphorus in the soil is modelled using first order kinetics and single-valued adsorption isotherms. Model details are provided in Appendix F.

ARM Model Calibration

Characteristics of the two watersheds for which ARM has been calibrated are listed in Table 6-1. Details of the calibration study are given in Donigian *et al.* (1977). Calibration results are summarized in Table 6-2 for the various water, soil and nutrient losses predicted by the model. In Table 6-2 "runoff" includes both surface runoff and interflow. In general, the results in Table 6-2 indicate that it is possible to calibrate the ARM model in such a way that it simulates dissolved nutrient losses with considerable accuracy. However, the losses of nutrients in solution are simulated much better than those associated with the sediment or eroded soil. The Georgia results are generally better than those for Michigan.

Cornell Nutrient Simulation (CNS) Model

The CNS model is described in detail in Appendix A. This model is a modification of an earlier model developed by Tubbs and Haith (1977). The model consists of three basic components: daily water balance, a daily soil loss (erosion) calculation, and monthly soil N and P inventories. The hydrologic model is based on the Soil Conservation Service's curve number

TABLE 6-1. WATERSHEDS USED IN ARM MODEL SIMULATIONS (Donigian et al. 1977)

Watershed	P2	P6
Location	Watkinsville, GA	East Lansing, MI
Soil Type	Cecil Sandy Loam	Spinks Loamy Sand
Area (ha)	1.3	0.8
Crop	Corn	Corn
Supporting Practices	Contoured	Contoured
Fertilizer Applications		
Nitrogen (kg/ha)	273	460
Phosphorus (kg/ha)	54	224
Calibration Period	5/74-9/75 exluding 2/75, 5/75	6/74-9/75 excluding 5/75

runoff equation, with suitable extensions to handle snowmelt. A daily soil moisture inventory is maintained for the top 30 cm of soil which is assumed to be homogeneous. Percolation is computed as any excess of soil water over field capacity in the 30-cm layer. Although soil water content is not modelled explicitly for depths below 30 cm, it is assumed that plant evapotranspiration needs in excess of the available water in the 30-cm surface layer will be met by soil water from lower depths. Soil erosion is estimated for each rainy day (exluding snowmelt periods in northern climates) using Onstad and Foster's (1975) modification of the universal soil loss equation. Daily runoff, percolation and erosion values are summed for each month and these monthly water and soil losses are inputs to the nutrient calculations.

Monthly inventories or mass balances are computed for soil N and P in the 30-cm surface layer. Although portions of plant nutrient needs may be supplied from nutrients (particularly N) at depths below 30 cm, N and P losses will be largely influenced by the levels of these nutrients in the surface layer. Separate organic and inorganic N inventories are maintained, but it is assumed that all soil inorganic N is converted to nitrate and hence moves readily in runoff and percolation waters. Runoff losses of solid-phase N are assumed to be organic nitrogen associated with eroded soil. Mineralization of organic N is considered temperature limited. Denitrification, ammonia volatilization from the soil and ammonium fixation are not modelled explicitly, but 25% of fertilizer N applications are subtracted to partially account for these loss mechanisms. Soil P is divided into available and fixed inorganic forms. The former is the pool of available or extractable P which can enter readily into soil solution. The partitioning of available P into dissolved and adsorbed

TABLE 6-2. COMPARISONS OF ARM MODEL SIMULATION RESULTS WITH OBSERVED LOSSES OF WATER, SEDIMENT AND NUTRIENTS (Donigian et al. 1977)

	Watershed P2 (Georgia)		Watershed P6 (Michigan)	
	Predicted	Observed	Predicted	Observed
Total Runoff (cm)	29	26	26	27
Sediment (MT/ha)	8.6	7.1	7.1	16.8
NO_3-N (kg/ha) in Runoff	2.5	1.5	6.3	2.2
NH_4-N (kg/ha) in Runoff				
Dissolved	2.8	2.2	2.0	2.8
Solid-Phase	2.8	2.3	0.2	----
Total N (kg/ha) in Runoff				
Dissolved	5.2	7.7	8.3	7.9
Solid-Phase	14.3	9.7	15.5	44.0
Dissolved (PO_4)-P (kg/ha)	0.4	0.3	2.3	1.4
Solid-Phase P (kg/ha)	10.1	5.5	8.1	23.3

constituents is based on a linear equilibrium isotherm which is a function of soil clay content and pH. In calculating losses of dissolved N and P, all inorganic N and dissolved available P in the 30-cm layer are considered available for loss in runoff. This assumption is necessary since the model cannot distinguish between subsurface and surface runoff. In situations where runoff consists entirely of overland flow, this assumption can lead to over-estimates of dissolved nutrient forms in runoff.

The CNS model has some disadvantages compared to ARM since it can only determine monthly nutrient losses and cannot provide estimates of losses of such nutrient forms as ammonium and organic N in solution. However, it does have several distinct advantages. The foremost feature of the model is that it is completely deterministic; i.e., all model parameters are based on the physical and climatic characteristics of the field being modelled and no calibration is needed. As a result, the CNS model can be applied to fields which have not been monitored for runoff and water quality data.

Validation of the CNS Model

A number of small watersheds (fields) were modelled in New York and Georgia to evaluate the accuracy of the model. Data for the validation were taken from two previous studies sponsored by the U.S. Environmental Protection Agency (Smith *et al*. 1978; Klausner *et al*. 1976).

The four Georgia watersheds (all at Watkinsville) are described in Table 6-3. All watersheds have a Cecil sandy loam soil. Although the watersheds were moldboard plowed at the beginning of the study, subsequent tillage practices on all watersheds were minimized (generally discing and harrowing). Watersheds P3 and P4 had major conservation practices (terracing, winter cover crops) as compared with P1 and P2 which, except for the final year on P1, had continuous row cropping with contouring only. The watersheds thus provided a test of the CNS model's capability to predict differences in the effects of conservation practices. Watersheds P2 and P4 were monitored for runoff, sediment and nutrient losses, while only runoff and sediment were measured on P1 and P3.

TABLE 6-3. SMALL WATERSHEDS AT WATKINSVILLE, GEORGIA, USED FOR CNS MODEL VALIDATION

Watershed		P1	P2	P3	P4
Area (ha)		2.7	1.3	1.3	1.4
Supporting Practice		Contouring	Contouring	Contouring, Terracing, Winter Cover	Contouring, Terracing, Winter Cover
Crop					
1972	Growing Season	Soybeans	----[a]	Soybeans	----[a]
	Winter Cover	None	----[a]	Rye	----[a]
1973	Growing Season	Soybeans	Corn	Soybeans	Corn
	Winter Cover	None	None	Rye	Rye
1974	Growing Season	Soybeans	Corn	Soybeans	Corn
	Winter Cover	Barley	None	Barley	Rye
1975	Growing Season	Sorghum	Corn	Soybeans	Corn

[a/] Watersheds P2 and P4 were not monitored in 1972.

Data for nutrient inputs, cropping dates and initial soil nutrient levels were obtained from Smith *et al*. (1978). Soil moisture parameters were obtained from soil surveys. Curve numbers and soil loss factors for the runoff and erosion submodels were obtained from standard sources (Ogrosky and Mockus, 1964; Wischmeier and Smith, 1978; Stewart *et al*. 1976). Model input parameters for the validation runs are summarized in Appendix A.

Validation results for the Georgia watersheds are given in Table 6-4. Model predictions and observations can be compared in several ways. Considering first the absolute prediction errors, it can be seen that model accuracy varies among watersheds. Considering the possible errors in both field measurements and model parameters, runoff and sediment predictions are quite accurate for watersheds P1 and P3. Sediment losses are substantially over-predicted for P2 and P4 and the runoff prediction is too small for P4. The large discrepancy between predicted and observed sediment losses for P4 may be due to sediment deposition in terrace channels and a grassed waterway. Such deposition was assumed not to occur in the model. Except for dissolved P losses from P4, all other nutrient losses are over-predicted. The errors in predicted solid-phase nutrient losses are due to the over-estimation of sediment losses, while the overprediction of dissolved N losses in runoff is likely due to the modelling assumption that all inorganic N in the top 30 cm of soil is available for runoff loss.

The significance of errors in predicted water, soil and nutrient losses depends on the purposes for which the predictions are made. In the current application, the ability of the simulation model to estimate the relative effectiveness of SWCPs is of most concern. Referring to Table 6-3, it can be seen that watersheds P1 and P3 have similar growing season crops, but different supporting practices. Although the two watersheds are not otherwise identical, it is reasonable to assume that supporting practices are the major factor accounting for differences in losses from the watersheds. A similar comparison is possible for watersheds P2 and P4. Observed and predicted runoff, sediment and nutrient losses from the two watershed pairs (P1, P3 and P2, P4) are indicative of the relative effectiveness of contouring (on P1 and P2) versus terracing and winter cover (on P3 and P4). If the predicted reductions in losses correspond to observations, this can provide at least a partial testing of the CNS model's ability to evaluate SWCPs.

Table 6-5 compares the predicted and observed reductions in water, sediment and nutrient losses achieved by the conservation practices on P3 and P4. These comparisons are generally encouraging. Predicted and observed reductions are all in the same direction (positive) and in most cases are of comparable magnitude. The significant exceptions are the effects of terracing on runoff and solid-phase N and dissolved P losses from watershed P4. These errors are mainly due to the underprediction of runoff and overprediction of sediment loss on P4. It appears that runoff and sediment loss for this watershed is not adequately accounted for by the parameters of the curve number runoff equation and the universal soil loss equation. Given the relatively shortness of the validation time period, this discrepancy is not surprising. A second factor may account for the small reduction in observed dissolved P losses in runoff. As noted in Section 4, the leaching of P from crop stubble and residues may be a significant source of this nutrient in runoff, and such leaching is not included in the simulation model.

A second validation of the CNS model was based on data from 24 field plots (each approximately 0.3 ha) at Aurora, N.Y. (Klausner, et al. 1976). The 24 plots represented three manure application rates, two management levels and three times of manure application. The lowest application rates were replicated, bringing the total to 24 plots. Straight-row corn was planted on all plots and the two management levels were "good" (retention of crop residues after harvest) and "poor" (removal of residues). The three manure application rates were 35,100 and 200 MT/ha of dairy manure per year. The predominant

TABLE 6-4. COMPARISONS OF CNS PREDICTIONS WITH OBSERVED LOSSES OF WATER, SEDIMENT AND NUTRIENTS FOR FOUR WATERSHEDS AT WATKINSVILLE, GEORGIA

	Runoff (cm)	Sediment (MT/ha)	Dissolved N in Runoff (kg/ha)	Solid-Phase N in Runoff (kg/ha)	Dissolved P in Runoff (kg/ha)	Solid-Phase P in Runoff (kg/ha)
Watershed P1[a]						
Predicted	52	62	--	--	--	--
Observed	62	64	--	--	--	--
Watershed P2[b]						
Predicted	36	32	22.0	16.6	0.8	7.8
Observed	48	19	9.6	9.2	0.6	5.8
Watershed P3[a]						
Predicted	36	13	--	--	--	--
Observed	48	12	--	--	--	--
Watershed P4[b]						
Predicted	20	18	8.3	10.6	0.2	3.1
Observed	42	6	5.1	3.6	0.5	1.6

[a] /July, 1972 - Sept., 1975; No nutrient observations available

[b] /Runoff and sediment data for May, 1973 - Sept., 1975; Nutrient data for May, 1974 - Sept. 1975

TABLE 6-5. COMPARISON BETWEEN CNS PREDICTED AND OBSERVED EFFECTS OF SWCPs

	% Reduction from P1 to P3		% Reduction from P2 to P4	
	Predicted	Observed	Predicted	Observed
Runoff	31	23	44	14
Sediment	79	81	44	68
Dissolved N in Runoff	--	--	62	47
Solid-Phase N in Runoff	--	--	36	61
Dissolved P in Runoff	--	--	75	17
Solid-Phase P in Runoff	--	--	60	72

soil on the plots was a moderately to poorly drained Lima-Kendaia silt loam. Runoff was collected in grass-lined interception ditches and percolation was measured by tile drainage on twelve of the plots.

Data for model validation was obtained from Klausner et al. (1976) and soil surveys. Additional, unpublished data for 1974 was provided by Stuart D. Klausner, Department of Agronomy, Cornell University. As with the Watkinsville validations, curve numbers for runoff calculations were obtained from Ogrosky and Mockus (1964). Model input data are summarized in Appendix A.

The Aurora validations provided additional tests of the CNS model which were not possible using the Watkinsville watersheds. In particular, the percolation and snowmelt components of the model could be tested at Aurora. In addition, since the Aurora study involved a large number of plots, model predictions and observed losses could be averaged in various ways among plots. The averaging is useful, since a single field or watershed is likely to include anomalies which are not accounted for in a simulation model.

The results of the Aurora validations are given in Table 6-6. Because of deposition in the grassed collection ditches, sediment loss data could not be considered reliable and hence were not used in validations. The predicted and observed losses are averaged over management and manure application rates. As with the Watkinsville results, runoff is under-predicted, but with significantly less error. Inorganic N losses are again over-predicted, but errors are again much less significant that at Watkinsville. This is likely due to the presence of greater portions of interflow in the Aurora runoff. The only significant errors in the validation are in the predictions of runoff for the poorly managed plots and of available P losses in runoff. The latter, however, is consistently 30-40% of the predicted value, indicating errors in estimates

TABLE 6-6. COMPARISON OF CNS PREDICTED AND OBSERVED LOSSES OF WATER, INORGANIC N AND AVAILABLE P FOR 24 0.3-ha FIELD PLOTS AT AURORA, N.Y. (1972-1974)

Plot Type	Runoff	Percolation	Dissolved N in Runoff	Dissolved N in Percolation	Dissolved P in Runoff
	-------(cm/yr)-------		---------------(kg/ha-yr)-----------------		
Well-Managed					
Predicted	5.1	45.2	4.8	55.7	0.22
Observed	5.7	32.2	3.9	45.6	0.55
Poorly Managed					
Predicted	6.3	43.4	6.5	57.2	0.30
Observed	13.2	45.3	5.5	70.4	0.83
35 MT/ha					
Predicted	5.7	44.3	4.3	40.0	0.14
Observed	6.1	41.8	2.7	43.5	0.31
100 MT/ha					
Predicted	5.7	----[a]	5.8	----[a]	0.28
Observed	16.1	----	8.4	----	0.91
200 MT/ha					
Predicted	5.7	44.3	8.2	72.9	0.49
Observed	9.4	35.6	5.0	72.6	1.23
Overall Average					
Predicted	5.7	44.3	5.6	56.4	0.26
Observed	9.4	38.8	4.7	58.0	0.69

[a] No percolation data available.

of available P in the soil or manure or an error in the computed value of the P adsorption coefficient. Both percolation and N losses in percolation are adequately predicted. The relative differences in losses observed among plots are also seen in the simulation predictions.

EVALUATION OF CONTOURING, TERRACES, SOD-BASED ROTATIONS AND CONSERVATION TILLAGE IN NEW YORK, IOWA AND GEORGIA

The effects of contouring, terraces, sod-based rotations and conservation tillage on water, soil and nutrient losses were simulated for Aurora, N.Y. (central New York), Tama, IA (central Iowa) and Watkinsville, GA (northeast Georgia). Nutrient, soil and water losses were estimated by 25 one-year (Jan. - Dec.) runs of the CNS model, using meteorological data for 1952-1976. Soil nutrient levels were reinitialized each year so that the results from each one-year run can be considered independent samples (assuming weather independence from year to year).

All SWCPs are compared with "conventional" management which is defined as continuous grain corn grown in straight rows with moldboard plowing and no supporting conservation practices. Soil and crop characteristics for conventional management are given in Table 6-7. Nutrient inputs and crop uptake for sod-based rotations are shown in Table 6-8.

The parameters given in Tables 6-7 and 6-8 were determined mainly from soil surveys and university extension staff in the relevant states (at Cornell, Iowa State and University of Georgia) and represent current cropping practice and soil properties at the respective locations. It should be noted that the properties of the New York soil would indicate greater susceptibility to runoff and erosion than the Iowa and Georgia soils. Average annual precipitation at the New York, Iowa and Georgia locations for the 25-year simulation period (1952-1976) was 88, 80 and 129 cm respectively. Soil characteristics are not changed for any SWCP. The only changes in crop characteristics for contouring and terracing (which is assumed to include contours) are appropriate adjustments in runoff curve numbers. The curve numbers for the average antecedent moisture conditions in New York, Iowa and Georgia are 84, 79 and 79 for contouring and 80, 74, 74 for terracing, respectively. Since the Iowa soil is assumed to be bare following fall plowing (except for snow), the curve number for fallow (86 for average antecedent moisture) is used during winter for conventional management, contouring, and terracing. This implies that contouring and terracing do not reduce runoff from bare soil during the winter.

Conservation tillage consists of chisel or shallow disc plowing in the spring at each location. Runoff curve numbers were chosen corresponding to "good" hydrologic condition (85, 78 and 78 for average antecedent moisture in New York, Iowa and Georgia, respectively). Sod-based rotations are assumed to be four years of corn followed by four years of alfalfa (and vice-versa) at all locations. Nutrient inputs and removals for these rotations are given in Table 6-8. For the purposes of curve number selection, it is assumed that the soil is in good hydrologic condition at all times. Fallow curve numbers were used in the nongrowing season for second, third and fourth year corn in Iowa.

TABLE 6-7. SOIL AND CROP CHARACTERISTICS FOR STUDY LOCATIONS

	New York	Iowa	Georgia
I. Soil Characteristics			
Type	Lima-Kendaia Silt Loam	Tama Silty Clay Loam	Cecil Sandy Loam
Hydrologic Group	C	B	B
Erodibility	0.30	0.28	0.28
Area (ha)	2	2	2
Slope (%)	5	3.5	4
Slope Length (m)	160	160	160
Bulk Density (g/cm^3)	1.31	1.22	1.65
Percent Clay	10	28	7
pH	6.8	6.7	7.0
Field Capacity (cm per 30 cm)	9.8	13.2	8.4
Wilting Point (cm per 30 cm)	4.9	6.6	4.2
Saturation (cm per 30 cm)	15.0	16.4	11.7
Organic N in top 10 cm (kg/ha)	2500	2450	600
Organic N Mineralization Rate (% per yr)	2	2	3
Initial Inorganic N in top 30 cm (kg/ha)	15	15	15
Phosphorus in top 30 cm (kg/ha)			
Available P	45	140	115
Total P	3100	2300	700

(cont'd) TABLE 6-7. SOIL AND CROP CHARACTERISTICS FOR STUDY LOCATIONS

	New York	Iowa	Georgia
II. Crop Characteristics (Conventional Management)			
Type	Continuous Corn	Continuous Corn	Continuous Corn
Emergence	June 1	May 15	May 15
Harvest	Oct. 31	Nov. 10	Nov. 1
Yield (kg/ha)	5020	7850	4080
Plowing	Spring Moldboard	Fall Moldboard	Spring Moldboard
Curve Number for Average Antecedent Moisture	88	81	81
Fertilizer N (kg/ha)	90	200	140
Fertilizer P (kg/ha)	35	80	35
Time of Fertilizer Application	Spring	Spring	Spring
Crop Uptake of N (kg/ha)	120	185	100
Crop Uptake of P (kg/ha)	20	30	20
Addition of Organic N from Crop Residues	40	65	35

The modelling of the effects of the legume (alfalfa) on the soil nitrogen budget requires a special set of assumptions. It is assumed that during the sod years the nitrogen fixed by the legume is not available for runoff or percolation losses. During the corn years, it is assumed that an inorganic N carryover from the legume would be available for crop growth and leaching losses (runoff and percolation). The assumed carryovers in the corn years are given in Table 6-8. The carryovers N is considered to be uniformly available from May through October, and was modelled as discrete inorganic N inputs during each of these months. The practical implication of these assumptions is that predicted dissolved N losses in runoff and percolation are very low for the legume years of the rotation.

The limited field data which are available appears to support the predictions for runoff, but perhaps not for percolation. Moderate levels of per-

TABLE 6-8. ANNUAL NUTRIENT INPUTS AND REMOVAL FOR ROTATIONS

Year/Crop	Inorganic N			Organic N Added in Residues	Available P	
	Fertilizer	Legume Carryover	Crop Uptake		Fertilizer	Crop Uptake
New York	kg/ha					
1. Corn	0	130	120	40	35	20
2. "	25	65	"	"	"	"
3. "	55	35	"	"	"	"
4. "	90	0	"	"	"	"
5. Hay	0	--	110	--	25	10
6-8. "	0	--	190	--	"	15
Iowa						
1. Corn	0	230	185	65	80	30
2. "	100	115	"	"	"	"
3. "	150	60	"	"	"	"
4. "	200	0	"	"	"	"
5. Hay	0	--	165	--	"	15
6-8. "	0	--	280	--	"	25
Georgia						
1. Corn	25	115	100	35	35	20
2. "	80	60	"	"	"	"
3. "	110	30	"	"	"	"
4. "	140	0	"	"	"	"
5. Hay	0	--	95	--	"	10
6-8. Hay	0	--	160	--	"	15

colation N losses have been observed for grass-legume mixtures and shallow-rooted legumes, particularly after being turned under in the fall (Kilmer, 1974). In the present modelling application, the sod crop is deep-rooted (alfalfa) and the release of fixed N is explicitly modelled as a carry over effect in the rotation years following the legume. Hence low predicted percolation N losses are not necessarily inconsistent with field data. Nevertheless, the uncertainty in the predictions must be recognized, since they have not been validated.

Cover factors (C in the universal soil loss equation) were selected for the appropriate crop stage from Wischmeier and Smith (1978). The conservation practice factor (P) for terraces is based in division of the slope length into 2 terraces. As suggested in Stewart et al. (1975) the resulting factors are multiplied by 0.2 to provide estimates of sediment loss. This assumes that 80% of the eroded soil is deposited in terrace channels. With all practices, it is assumed that crop residues are not removed after harvest.

Mean Annual Water and Sediment Losses

Mean annual runoff, percolation and sediment losses obtained from the 25-year simulations are shown in Table 6-9. The effects of contouring, terracing and sod rotations in controlling sediment are similar for the three locations, but conservation tillage effects are more site-specific. The effects of all SWCPs on water movement are highly site-specific. For example, terraces reduce runoff by 60, 30 and 40% at the New York, Iowa and Georgia sites, respectively. The high runoff reductions for the poorly drained New York soil suggests that SWCPs may be relatively more effective on soils which are highly susceptible to runoff. All practices except sod rotations are significantly more effective in controlling sediment than runoff. One effect of runoff reduction is increased percolation, although the increase is not always equal to the runoff reduction. In Iowa, runoff reduction may make more water available in the upper soil layer for evapotranspiration and hence only a portion of the additional infiltration percolates.

Average Annual Nutrient Losses

Mean annual nutrient losses obtained from the simulation runs are shown in Table 6-10. Probably the most significant aspect of the table is that the four SWCPs reduce every category of nutrient loss with the exception of N in percolation. Moreover the percentage increases in percolation N are small compared to reductions in runoff N. One of the practices, sod-based rotations, achieves substantial reductions in all loss categories, since it reduces runoff, soil loss and fertilizer applications.

In comparison with conventional management, conservation tillage is less effective at reducing solid-phase nutrient losses in New York and Georgia than in Iowa. This is because the soil is protected with residues during the dormant season with all practices at the former locations, while corn residues are plowed under in the fall with all practices except for conservation tillage in Iowa.

TABLE 6-9. MEAN ANNUAL RUNOFF, PERCOLATION AND SEDIMENT LOSSES FOR SELECTED SWCPs

Practice	Runoff		Percolation		Sediment	
	Mean (cm)	%[a] Change	Mean (cm)	% Change	Mean (MT/ha)	% Change
New York						
Conventional	10	---	39	---	20	---
Contouring	6	-40	43	+10	7	-65
Terracing	4	-60	45	+15	1	-95
Sod Rotation	3	-70	42	+10	6	-70
Conservation Tillage	8	-20	41	+ 5	9	-55
Iowa						
Conventional	7	---	25	---	17	---
Contouring	6	-15	26	+ 5	8	-55
Terracing	5	-30	26	+ 5	1	-95
Sod Rotation	3	-55	27	+10	7	-60
Conservation Tillage	5	-30	27	+10	5	-70
Georgia						
Conventional	19	---	58	---	41	---
Contouring	13	-30	64	+10	17	-60
Terracing	11	-40	66	+15	2	-95
Sod Rotation	13	-30	64	+10	16	-60
Conservation Tillage	16	-15	61	+ 5	24	-40

[a] Rounded to nearest 5%

TABLE 6-10. MEAN ANNUAL NUTRIENT LOSSES FOR SELECTED SWCPs

Practice	Dissolved N in Runoff		Dissolved P in Runoff		Dissolved N in Percolation		Solid-Phase N in Runoff		Solid-Phase in Runoff	
	Mean (kg/ha)	%[a] Change	Mean (kg/ha)	% Change	Mean (kg/ha)	% Change	Mean (kg/ha)	% Change	Mean (kg/ha)	% Change
New York										
Conventional	10	---	0.15	---	41	---	93	---	33	---
Contouring	5	-50	0.08	-45	45	+10	34	-65	12	-65
Terracing	3	-70	0.06	-60	47	+15	4	-95	1	-95
Sod Rotation	2	-80	0.04	-75	30	-25	29	-70	10	-70
Conservation Tillage	7	-30	0.11	-25	44	+ 5	42	-55	15	-55
Iowa										
Conventional	16	---	0.24	---	59	---	82	---	23	---
Contouring	12	-25	0.19	-20	64	+10	37	-55	10	-55
Terracing	11	-30	0.17	-30	65	+10	5	-95	1	-95
Sod Rotation	4	-75	0.11	-55	30	-50	33	-60	9	-60
Conservation Tillage	11	-30	0.16	-35	65	+10	24	-70	7	-70
Georgia										
Conventional	14	---	0.39	---	51	---	41	---	17	---
Contouring	9	-35	0.28	-30	56	+10	17	-60	7	-60
Terracing	7	-50	0.23	-40	59	+15	2	-95	1	-95
Sod Rotation	6	-55	0.28	-30	34	-35	16	-60	6	-65
Conservation Tillage	12	-15	0.32	-20	54	+ 5	24	-40	10	-40

---

[a] Rounded to nearest 5%

The major nutrient losses are dissolved N in percolation and solid-phase N and P in runoff. The solid-phase losses are sharply reduced by all SWCPs, following the reduction of sediment loss. The reductions in dissolved N and P in runoff from most SWCPs are significantly less than reductions in particulate P and N. The exceptions are sod-based rotations which are equally effective at controlling losses of nutrients associated with sediment and those which are in solution with the runoff.

The reductions in runoff losses of dissolved nitrogen are almost exactly balanced by increases in percolation N for contouring, terracing and conservation tillage. The significance of these percolation losses is conjectural. If percolation waters subsequently emerge as base flow in streams, and the leached nitrogen is conserved without dilution in the groundwater, these three practices may not reduce stream nitrogen loadings in the long run. However, it is unlikely that all the N leached from the top 30 cm of soil (as predicted by the simulation model) will reach either groundwater aquifers or streams. Substantial portions of the dissolved N which moves below 30 cm can and will be taken up by plant roots, particularly corn and alfalfa roots. Therefore, the 5-15% increases in percolation N losses probably should not be considered significant.

Yearly Variations in Nutrient Losses

As noted in the introduction to this chapter, the effects of SWCPs can be expected to vary from year to year, primarily due to weather conditions. These variations can be considerable, as indicated by the simulation results shown in Table 6-11. Annual losses of dissolved available P and N in runoff are compared or the first 15 years of CNS model runs for Iowa. As is apparent, the effectiveness of contouring in controlling these nutrient losses is highly variable (0-69% for N, 0-50% for P). Moreover the relative effectiveness of P and N control is not consistent (e.g., in 1953, N in controlled much more than P, but in 1957 both are controlled equally).

These results suggest that SWCPs cannot be evaluated based only on two or three years of data. This is equally true whether the nutrient loss data are obtained from field measurements or simulation models. In either case, studies must be of sufficient duration to produce the full expected range of possible nutrient losses.

While mean annual losses are probably the most meaningful results which can be obtained from the 25-yr simulations, other statistical parameters can provide insight to the relative effectiveness of SWCPs. One of these parameters is the coefficient of variation of annual nutrient losses (standard deviation of annual losses divided by mean annual losses). These coefficients are summarized in Table 6-12. High coefficients indicate that the nutrient losses associated with a practice vary substantially from year to year. It can be noted from Table 6-12 that all the SWCPs studied tend to increase the relative variability of dissolved nutrients in runoff, primarily due to the practices' non-linear effects on runoff. Small runoff events are often prevented completely by SWCPs while much larger events are only partially controlled. This increase in a variability is not seen in solid-phase nutrient losses, since rainfall erosivity remains the same for all practices. Dissolved

TABLE 6-11. ESTIMATED EFFECTIVENESS OF CONTOURING IN CONTROLLING NUTRIENT LOSSES OVER 15 YEARS IN IOWA

	Dissolved N in Runoff			Dissolved P in Runoff		
Year	Conventional (kg/ha)	Contoured (kg/ha)	% Reduction	Conventional (kg/ha)	Contoured (kg/ha)	% Reduction
1952	4.8	1.5	69	0.12	0.06	50
1953	5.6	3.9	30	0.13	0.12	8
1954	21.4	13.6	36	0.43	0.35	19
1955	16.2	11.7	28	0.15	0.12	20
1956	1.2	1.2	0	0.01	0.01	0
1957	21.0	16.3	22	0.27	0.21	22
1958	12.6	8.2	35	0.14	0.09	36
1959	17.3	12.2	29	0.23	0.18	22
1960	31.4	26.3	16	0.41	0.35	15
1961	13.8	9.0	35	0.27	0.20	26
1962	8.2	3.9	52	0.08	0.04	50
1963	28.6	24.3	15	0.35	0.30	14
1964	26.9	22.4	17	0.28	0.25	11
1965	19.5	15.7	19	0.29	0.25	14
1966	24.9	17.1	31	0.26	0.19	27

N losses in percolation are much less variable from year to year, and except for sod-based rotations SWCPs have little effect on coefficients of variation.

With the exception of dissolved P the variation in annual nutrient losses associated with sod-based rotations significantly exceeds that of other SWCPs. This is of course due to the large differences in dissolved N and sediment losses between the sod and corn years in the rotation. Thus the practice alternates (sod) years of very effective nutrient control with (corn) years in which nutrient losses are only moderately lower than those associated with conventional management. Although nutrient losses averaged over the eight-year rotation are relatively low, sod-based rotations are of limited effectiveness during the latter two years of the four-year corn sequence.

The results shown in Table 6-12 suggest that SWCPs are relatively less effective at controlling dissolved nutrient losses in extreme or wet years (i.e., years characterized by large runoff events) than they are at controlling average annual losses. This conclusion can be demonstrated by examining the probability distribution functions (cumulative probabilities) of annual losses. As an example, these functions are shown in Figures 6-1, 6-2, 6-3 for losses of dissolved P associated with three practices. If a wet or extreme year is defined as a year in which nutrient losses are exceeded only one year in ten, then the extreme losses can be approximated as the 90% values from the probability distributions. These losses are summarized in Table 6-13, which compares the effectiveness of two SWCPs in controlling average and one-in-ten year losses of dissolved P in runoff. It can be seen that both contouring and sod-based rotations offer less reduction of the one-in-ten year dissolved P losses than of average annual losses, although the differences in percentage reductions are relatively small for contouring.

The probability distributions can be used to compare practices in several additional ways. One evaluation is a comparison of the probabilities that losses will be kept below a certain level. For example, in New York, with conventional management, there is only a 33% probability that dissolved P losses in any year will not exceed 0.10 kg/ha. The probability increases to 69% and 92%, respectively, for contouring and sod rotations. The converse to these probabilities is an assessment of risk. Thus the risk of dissolved P loss greater than 0.10 kg/ha in any year decreases from 67% to 8% with a sod rotation.

EVALUATION OF REDUCED TILLAGE, CONTOURING, TERRACING AND NO-TILLAGE FOR CALIBRATED GEORGIA AND MICHIGAN WATERSHEDS

The ARM simulation model was used to estimate the effects of tillage management practices on nutrient losses. Unlike the CNS model, ARM contains several parameters that are directly related to tillage. Since the ARM calibrations were for reduced or minimum tillage conditions, the moldboard plowing of the previous section's "conventional management" could not be duplicated. Evaluations of the following four conditions were possible.

1. Reduced Tillage - (Base Condition) Row crops in straight rows parallel to the land slope. Tillage of the soil is in the form of discing and is done in the spring to prepare the soil for planting. Afer har-

TABLE 6-12. COEFFICIENTS OF VARIATION OF ANNUAL NUTRIENT LOSSES

Practice	Dissolved N in Runoff	Dissolved P in Runoff	Dissolved N in Percolation	Solid-Phase N in Runoff	Solid-Phase P in Runoff
			kg/ha-yr		
New York					
Conventional	0.5	0.5	0.2	0.6	0.6
Contouring	0.7	0.6	0.1	0.6	0.6
Terracing	0.8	0.7	0.1	0.6	0.6
Sod Rotation	1.1	0.8	0.6	1.1	1.1
Conservation Tillage	0.6	0.5	0.1	0.6	0.6
Iowa					
Conventional	0.5	0.5	0.3	0.5	0.5
Contouring	0.6	0.5	0.3	0.5	0.5
Terracing	0.7	0.5	0.3	0.5	0.5
Sod Rotation	1.4	0.7	0.8	1.4	1.4
Conservation Tillage	0.7	0.6	0.2	0.5	0.5
Georgia					
Conventional	0.5	0.5	0.2	0.6	0.6
Contouring	0.7	0.7	0.2	0.6	0.6
Terracing	0.8	0.8	0.2	0.6	0.6
Sod Rotation	1.0	0.7	0.6	1.3	1.4
Conservation Tillage	0.6	0.6	0.2	0.5	0.5

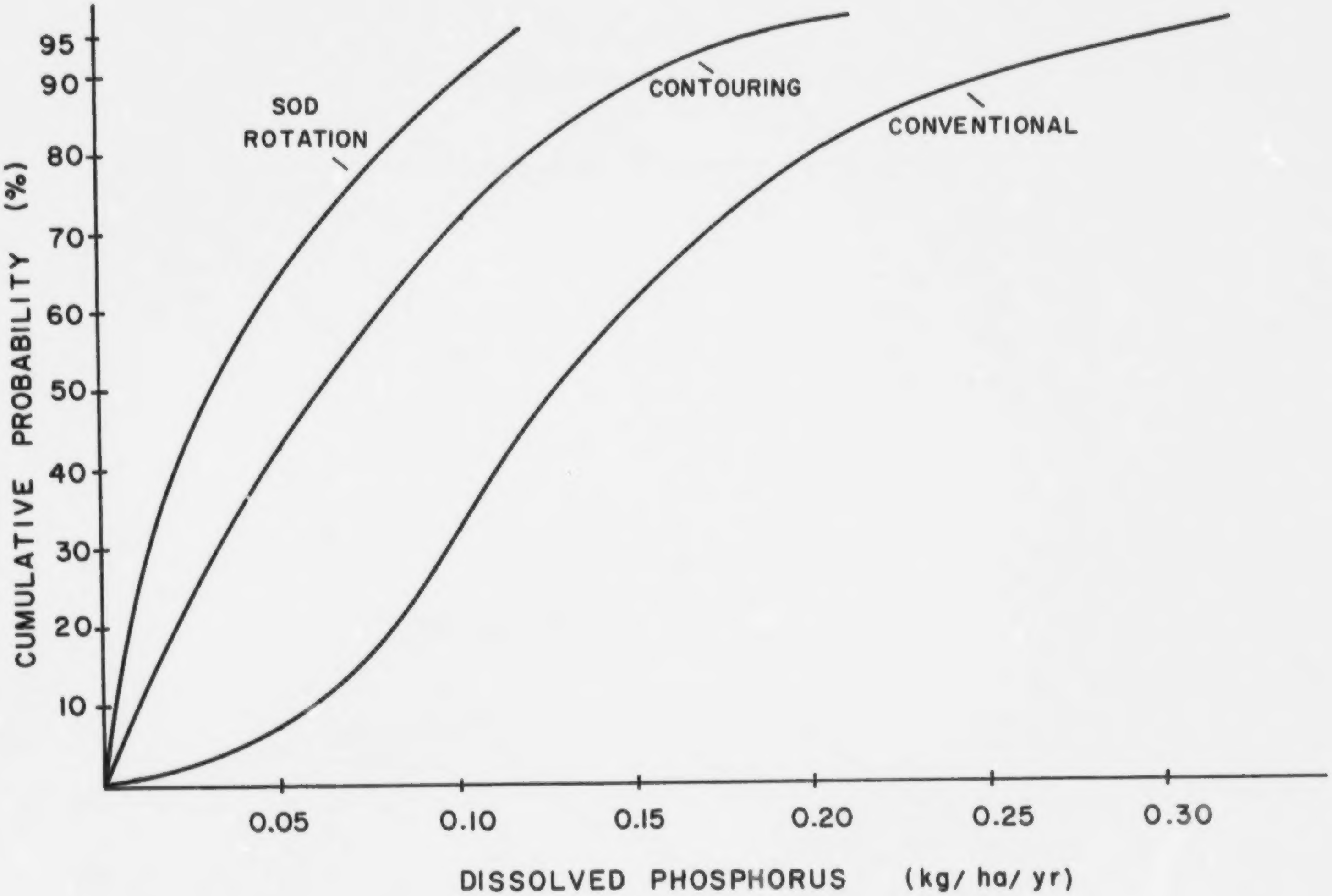

FIGURE 6-1. PROBABILITY DISTRIBUTION FUNCTIONS FOR DISSOLVED PHOSPHORUS IN RUNOFF - NEW YORK.

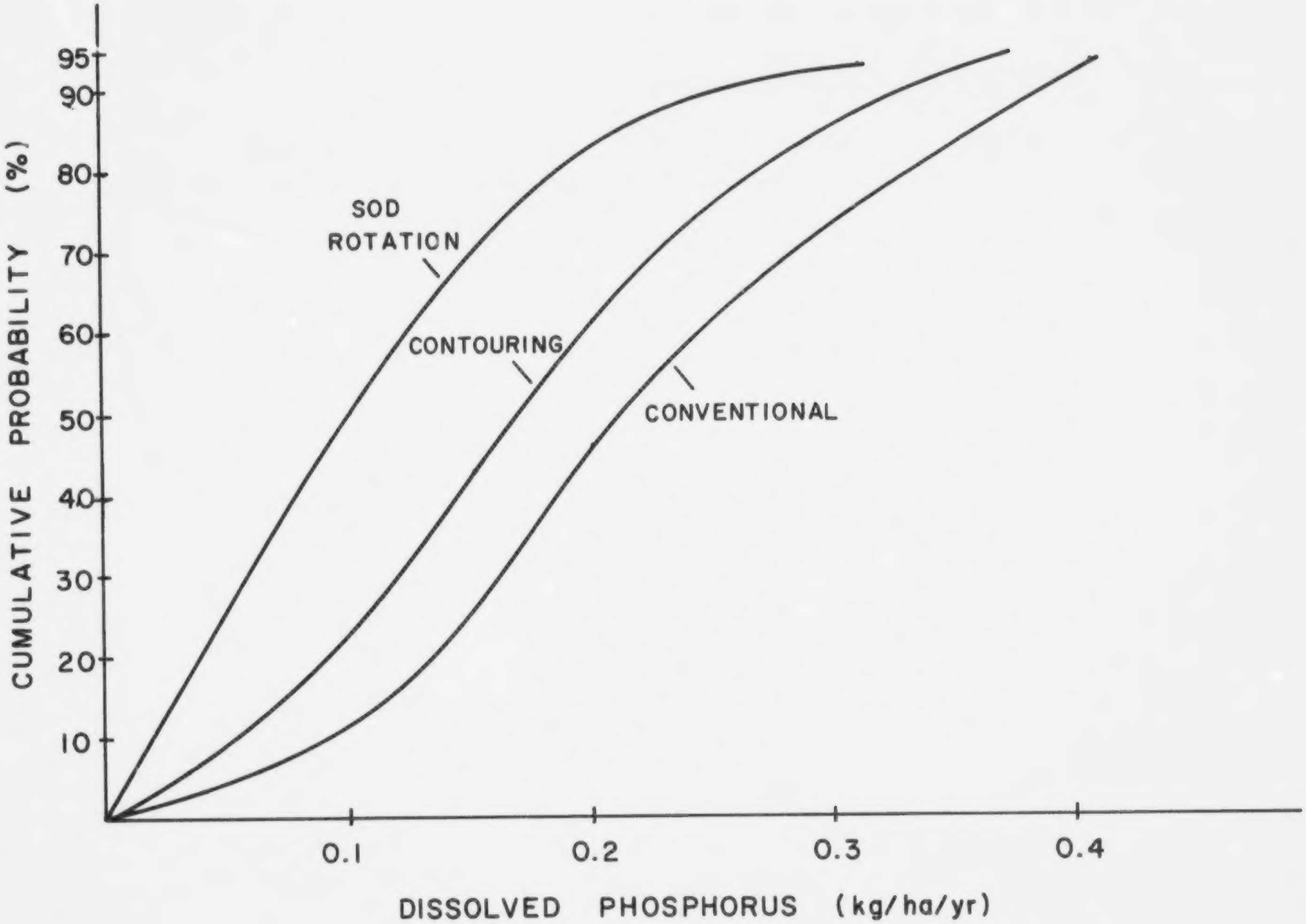

FIGURE 6-2. PROBABILITY DISTRIBUTION FUNCTIONS FOR DISSOLVED AVAILABLE PHOSPHORUS IN RUNOFF - IOWA.

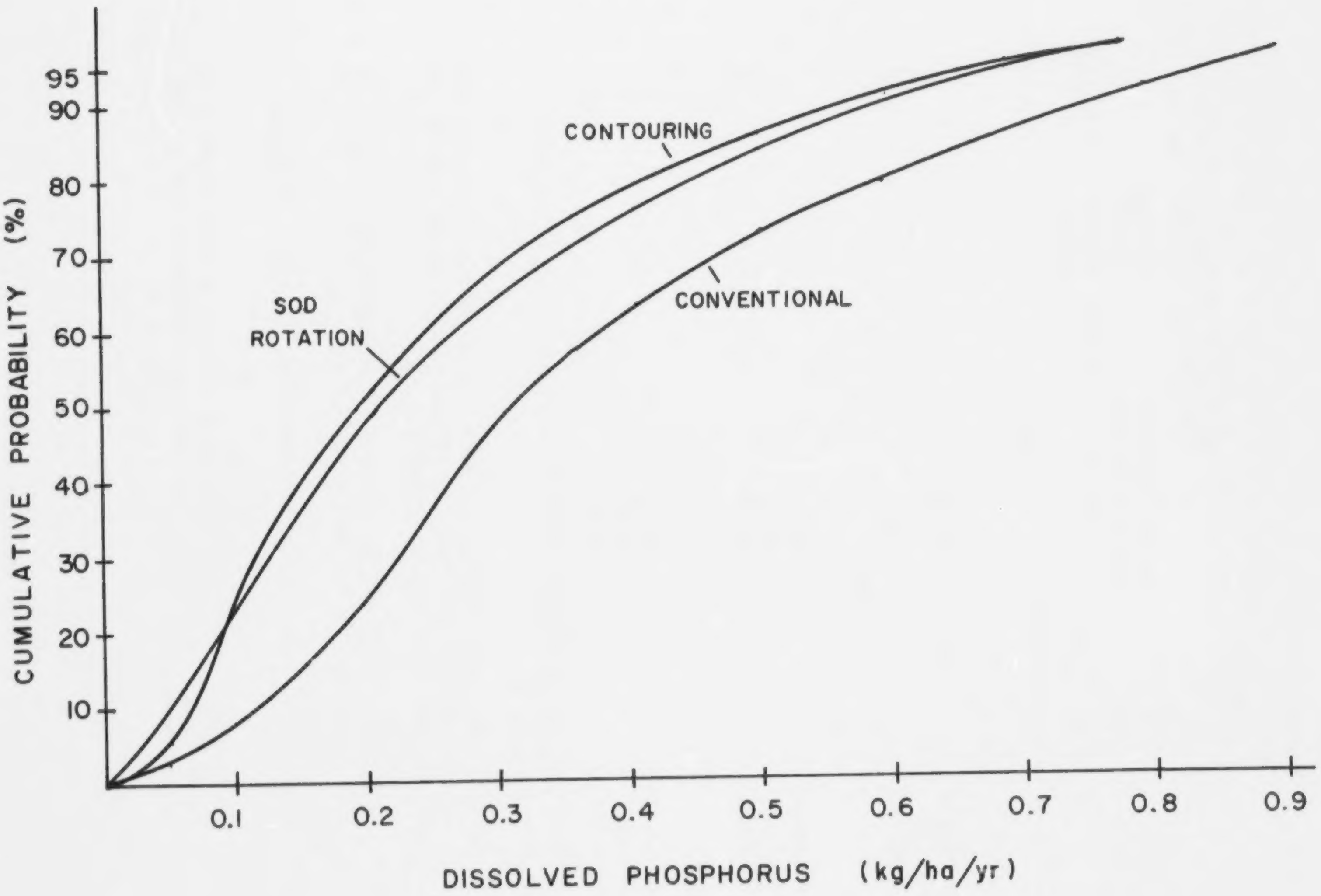

FIGURE 6-3. PROBABILITY DISTRIBUTION FUNCTIONS FOR DISSOLVED AVAILABLE PHOSPHORUS IN RUNOFF - GEORGIA

TABLE 6-13. COMPARISON OF THE EFFECTIVENESS OF CONTOURING AND SOD-BASED ROTATIONS IN CONTROLLING AVERAGE AND EXTREME LOSSES OF DISSOLVED P IN RUNOFF

	Dissolved P in Runoff			
Practice	Losses Exceeded One Year in Ten (kg/ha-yr)	% Change	Average Annual Losses (kg/ha-yr)	% Change
New York				
Conventional	0.26		0.15	
Contouring	0.16	-40	0.08	-45
Sod Rotation	0.10	-60	0.04	-75
Iowa				
Conventional	0.38		0.24	
Contouring	0.32	-15	0.19	-20
Sod Rotation	0.25	-35	0.11	-55
Georgia				
Conventional	0.78		0.39	
Contouring	0.64	-20	0.28	-30
Sod Rotation	0.67	-15	0.28	-30

[a] Rounded to nearest 5%

vest, the crop residue is removed from the field and the soil is fallow through the winter. This situation is defined as "base conditions" as described in Appendix F.

2. Contours - Identical to reduced tillage with the exception that rows are perpendicular to land slope.

3. Terraces - Identical to contours with the watershed divided into two terraces.

4. No-Tillage - Differs from reduced tillage in that there is no tillage prior to spring planting and the crop residue is left on the field after harvest.

Reduced tillage is essentially minimum or conservation tillage without crop residues. The remaining conditions impose additional supporting practices: contouring, terraces, tillage elimination and residue retention. Each of the four alternatives was simulated using 10-year ARM model runs for watersheds P2 and P6 in Georgia and Michigan, respectively (Table 6-1). These simulations are described in detail in Appendix F. The results, which are summarized in Tables 6-14 and 6-15, indicate the changes in nutrient losses which can be achieved by contouring, terracing and no-tillage as compared to tillage reduction alone. It should be noted that these results are not directly comparable to those obtained from the CNS simulations described in the preceding section. Different watersheds and locations were simulated using CNS and the standards of comparison for the two simulations are different ("conventional management," implying moldboard plowing for CNS and reduced tillage or spring discing for ARM).

The effects of contouring, terracing and no-tillage on runoff and sediment losses as predicted by the ARM model, are compared with reduced tillage in Table 6-14. Runoff predictions are divided into two components, overland flow and interflow. Contouring and no-tillage provide only minimal decreases in runoff compared to the reduced tillage base conditions, but substantial reductions in runoff are achieved by the addition of terraces. All practices significantly reduce sediment loss, although the relative effectiveness of the SWCPs is different for the Georgia and Michigan watersheds. For watershed P6 (Michigan), the no-tillage practice is much more effective than countouring or terraces, while in Georgia watershed P2 the effects of all practices are somewhat similar.

The effects of the SWCPs on nutrient losses are shown in Table 6-15. The practices all offer substantial control of solid-phase nutrients which travel with the sediment. The most striking aspects of the simulations are the minimal or nonexistant reductions in losses of dissolved nutrients in runoff with contouring, terracing and no-tillage when compared to the effects of reduced tillage alone. The increase in dissolved nutrients with no-tillage can be partially explained by the nutrient additions from residues. The absence of significant reductions in dissolved N losses in runoff may be due to increased interflow, which leaches nitrate from the soil.

A general conclusion which can be made from these results is that the addition of either contouring or terraces to tillage reductions is substantially more effective at reducing particulate nutrient losses associated with sediment than dissolved losses in runoff. No-tillage will also reduce particulate nutrient losses, but reduces neither dissolved N and available P in runoff.

COMPARISON OF CNS AND ARM SIMULATION PREDICTIONS

The CNS and ARM simulation results reported in the previous two portions of this section cannot be directly compared since the locations, practices and simulation periods were different for the two models. In an attempt to evaluate the consistency of results produced by the two models, the CNS model was used to simulate the same practices which were modelled by ARM (reduced tillage, contouring, terracing and no-tillage) for the Georgia watershed P2. Unfortunately, there are limitations on CNS's ability to model these practices. The selection of curve numbers was somewhat arbitrary. The soil was assumed to be in poor hydrologic condition for reduced tillage, contouring and terracing. Fallow curve numbers for straight row cropping were used between harvest and planting for these three SWCPs, since comparable numbers are unknown for contouring and terracing. Curve numbers for good hydrologic condi-

TABLE 6-14. MEAN ANNUAL RUNOFF AND SEDIMENT LOSSES FOR SELECTED SWCPs AS PREDICTED BY ARM MODEL

Practice	Runoff				Sediment Loss	
	Overland Flow (cm)	Interflow (cm)	Total (cm)	% Change in Total[a]	Mean (MT/ha)	% Change
Watershed P2						
(Georgia)						
Reduced Tillage[b]	18.6	1.3	19.9	---	6.3	---
Contouring[b]	16.4	1.4	17.8	-10	4.0	-35
Terracing[b]	11.1	1.7	12.8	-35	2.8	-55
No-Tillage[c]	17.1	1.5	18.6	- 5	3.3	-50
Watershed P6						
(Michigan)						
Reduced Tillage[b]	16.5	2.6	19.1	---	3.5	---
Contouring[b]	15.4	2.7	18.1	- 5	2.8	-20
Terracing[b]	11.2	3.2	14.4	-25	2.4	-30
No-Tillage[c]	15.6	2.7	18.3	- 5	1.0	-70

a/ rounded to nearest 5%
b/ Spring discing, residues removed
c/ Residues left

TABLE 6-15. MEAN ANNUAL NUTRIENT LOSSES FOR SELECTED SWCPs AS PREDICTED BY ARM MODEL

Practice	Dissolved N in Runoff		Dissolved P in Runoff		Solid-Phase N in Runoff (Organic N + Adsorbed NH_4-N)		Solid-Phase PO_4-P in Runoff	
	(kg/ha)	% Change[a]	(kg/ha)	% Change	(kg/ha)	% Change	(kg/ha)	% Change
Watershed P2								
(Georgia)								
Reduced Tillage[b]	5.0	---	0.48	---	8.5	---	2.5	---
Contouring[b]	5.1	0	0.47	- 0	6.4	-25	1.9	-25
Terracing[b]	5.9	+20	0.46	- 5	5.0	-40	1.4	-45
No-Tillage[c]	5.6	+10	0.56	+15	7.5	-10	1.6	-35
Watershed P6								
(Michigan)								
Reduced Tillage[b]	16.9	---	6.6	---	4.3	---	1.6	---
Contouring[b]	15.7	- 5	6.0	-10	3.7	-15	1.3	-20
Terracing[b]	16.4	- 5	6.0	-10	3.2	-25	1.2	-25
No-Tillage[c]	18.1	+ 5	6.7	0	2.2	-50	0.6	-65

a/Rounded to nearest 5%

b/Spring discing, residues removed

c/Residues left

tions were used year-round for no-tillage. Simulations were based on the same ten years of weather data which were used in the ARM simulations. The results of the CNS simulations are compared in Tables 6-16 and 6-17 with those obtained previously from ARM.

Runoff and Sediment Predictions (Table 6-16)

Runoff

The magnitudes of runoff predictions are comparable for both models but the relative effects of terracing and no-tillage are reversed. The small percent change for terracing predicted by CNS is a consequence of using straight-row curve numbers during the non-growing season. This is probably incorrect. Conversely, the minimal reduction in runoff predicted by the ARM for no-tillage is inconsistent with the literature results reported in Section 4.

Sediment

Sediment predictions differ by a factor of three, even though sediment predictions for the watershed were well calibrated for ARM (Table 6-2) and validated for CNS (Table 6-4). However, the sediment losses predicted by ARM appear much too low, particularly when compared with observed losses (Table 6-4). The discrepancy is inexplicable. However, the predicted relative effects of SWCPs are very similar for both models.

Nutrient Predictions (Table 6-17)

Solid-Phase N and P

The predicted N and P losses from CNS are substantially greater than those obtained from ARM except for no-tillage. The percent reductions in these losses due to the SWCPs are somewhat greater with CNS, since the solid-phase losses are assumed to be proportional to sediment. Based on the earlier calibration and validation results, the CNS model predicts particulate nutrient losses with better accuracy that ARM.

Dissolved P

Predictions of dissolved P losses are very similar for reduced tillage, contouring and terracing. The no-tillage results differ substantially. Since, as noted earlier, the CNS model does not consider the phosphorus dynamics associated with plant residues, and the ARM model has not been calibrated for no-tillage, the accuracy of predictions of dissolved P losses for no-tillage is questionable for both models.

Dissolved N

Perhaps the most interesting and significant differences in the two sets of model predictions are in losses of dissolved N in runoff. The magnitudes of N losses are very different, but this is due to the CNS model's consistent over-prediction of these losses. Given the potential errors in both models, percentage changes for contouring cannot be considered different. However, ARM predicts that terraces and no-tillage will increase dissolved N in runoff while CNS predicts reductions. These differences are important, since they lead to different conclusions regarding the effectiveness of these SWCPs. Most fundamentally, the CNS results imply that reductions in runoff decrease dissolved N losses, but ARM predicts that runoff reductions increase these losses.

TABLE 6-16. COMPARISON OF ARM AND CNS RUNOFF AND SEDIMENT PREDICTIONS - WATERSHED P2

	Total Runoff				Sediment Loss			
	ARM		CNS		ARM		CNS	
	(cm/yr)	% Change[a]	(cm/yr)	% Change	(MT/ha-yr)	% Change	(MT/ha-yr)	% Change
Reduced Tillage[b]	19.9	---	22.2	---	6.3	---	24.4	---
Contouring[b]	17.8	-10	21.4	- 5	4.0	-40	14.8	-40
Terracing[b]	12.8	-35	20.8	- 5	2.8	-60	9.6	-60
No-Tillage[c]	18.6	- 5	14.6	-35	3.3	-50	8.0	-65

a/Rounded to nearest 5%

b/Spring discing, residues removed

c/Residues left

TABLE 6-17. COMPARISON OF ARM AND CNS NUTRIENT LOSS PREDICTIONS - WATERSHED P2

	Solid-Phase N in Runoff				Solid-Phase P in Runoff			
	ARM		CNS		ARM		CNS	
	(kg/ha-yr)	% Change[a]	(kg/ha-yr)	% Change	(kg/ha-yr)	% Change	(kg/ha-yr)	% Change
Reduced Tillage[b]	8.5	---	16.1	---	2.5	---	9.0	---
Contouring[b]	6.4	-25	9.6	-40	1.9	-25	5.3	-40
Terracing[b]	5.0	-40	6.5	-60	1.4	-45	3.5	-60
No-Tillage[c]	7.5	-10	5.4	-65	1.6	-35	3.0	-65

	Dissolved P in Runoff				Dissolved N in Runoff			
	ARM		CNS		ARM		CNS	
	(kg/ha-yr)	% Change	(kg/ha-yr)	% Change	(kg/ha-yr)	% Change	(kg/ha-yr)	% Change
Reduced Tillage[b]	0.48	---	0.57	---	5.0	---	14.6	---
Contouring[b]	0.47	0	0.55	- 5	5.1	+ 0	13.3	-10
Terracing[b]	0.46	- 5	0.52	-10	5.9	+20	10.8	-25
No-Tillage[c]	0.56	+15	0.38	-35	5.6	+10	10.3	-30

[a] Rounded to nearest 5%

[b] Spring discing, residues removed

[c] residues left

There may be several reasons for these differences, but the most significant may be the handling of subsurface flows in the two models. With ARM, a portion of the subsurface drainage water appears as interflow in runoff. Increases in interflow, as with terracing and no-tillage (Table 6-14) will leach additional soil N which is reflected in increases in runoff N. Conversely, interflow is not considered in the CNS model, and runoff reductions are reflected in comparable increases in percolation. The effect of this is shown in Table 6-18. Considering the total dissolved N loss in runoff and percolation water, it can be seen that this total is not reduced by contouring, terracing or no-tillage compared to reduced tillage.

TABLE 6-18. DISSOLVED N LOSSES FROM WATERSHED P2 PREDICTED BY CNS MODEL

	Dissolved N Loss					
	Runoff		Percolation		Total	
	(kg/ha-yr)	% Change[a]	(kg/ha-yr)	% Change	(kg/ha-yr)	% Change
Reduced Tillage	14.6	---	47.4	---	62.0	---
Contouring	13.3	-10	48.7	+ 5	62.0	0
Terracing	10.8	-25	51.3	+10	62.1	0
No-Tillage	10.3	-30	53.6	+15	63.9	+ 5

[a] Rounded to nearest 5%

Summary of Model Comparison

In some ways this comparison of the two simulation models is an academic excercise. The ARM model has not been validated and the CNS model is not well-suited for predicting the effects of alternative tillage practices. Nevertheless, the excercise is informative. Although it does not appear that any meaningful comparison of no-tillage is possible, the results for reduced tillage, contouring and terracing yield several conclusions.

1. As indicated by the magnitudes of runoff, sediment and nutrient losses, the two models produce significantly different results.

2. In spite of the absolute differences noted above, the relative effects of adding contouring or terracing to a reduced tillage base condition are consistent with both models for sediment, particulate nutrients and dissolved P. Thus both models predict that for this watershed contouring or terraces will substantially reduce losses of sediment and particulate nutrients but will have negligible effect on dissolved P losses.

3. The substantial difference in model predictions of the effect of terracing on dissolved N losses is due to assumptions concerning subsurface water movement. The partioning of subsurface drainage into percolation and interflow can significantly affect the quantities of dissolved N in runoff.

SUMMARY AND CONCLUSIONS

The results of these simulation studies must be interpreted with care. The ranking of SWCPs based on relatively small differences in predicted nutrient losses is clearly inappropriate. The basic value of the modelling results is the demonstration of consistent patterns of results. The conclusions presented in this section are based mainly on predictions of nutrient losses obtained from the 25-year runs of the Cornell Nutrient Simulation (CNS) model for comparable 2-hectare fields in New York, Iowa and Georgia. Although the CNS model was found to have predictive errors, validation results indicated that it is generally capable of estimating the relative effects of management practices. The conclusions apply to corn grain cropping under five conditions: conventional management (moldboard plowing, straight-row cropping), contouring, terraces, sod-based rotations and conservation tillage. SWCPs were all compared with conventional management. The corn stalk residues are left on the field under all conditions. Sod-based rotations are four consecutive years each of corn and alfalfa. Conservation tillage is shallow disc or chisel spring plowing.

The principal conclusions are as follows:

1. All SWCPs reduce all forms of nutrient loss except for dissolved N in percolation.

2. The effects of contouring, terracing and sod-based rotations on solid-phase nutrient losses are similar for the three locations. The effects of these practices on dissolved N and P in runoff and dissolved N in percolation varies significantly with location and can be considered site-specific. Conservation tillage is site-specific for all nutrient forms.

3. Sod-based rotations are unique in their ability to significantly reduce losses of dissolved and solid-phase nutrients in runoff and dissolved nutrients in percolation. Contouring, terracing and conservation tillage are substantially more effective at reducing runoff losses of nutrients in the solid-phase form than in the dissolved form. These three practices also increase percolation losses of dissolved N.

4. The effectiveness of conservation tillage depends on the prevailing management practice which it replaces. In Iowa, conventional management was defined as fall moldboard plowing, which leaves the soil bare during the non-growing period. Conservation tillage provides a residue cover during this period and is hence fairly effective in reducing dissolved N and P losses in runoff. Conversely, since spring plowing was considered typical in New York and Georgia, residue cover was equivalent for both conventional management and conservation tillage and hence the SWCP was relatively less effective for these two locations.

5. Losses of all nutrient forms vary significantly from year to year. More importantly, the effects of SWCPs show similar yearly variations.

ACKNOWLEDGMENTS

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SECTION 7

SIMULATION OF THE ACTION OF SOIL AND WATER CONSERVATION PRACTICES IN CONTROLLING PESTICIDES

Tammo S. Steenhuis

Various studies have examined the effects of certain SWCPs on losses of certain pesticides. Triplett et al. (1978) found that losses of atrazine and cyanazine were less under no-till than under conventional tillage. Baker and co-workers (1977, 1978) compared ridge planting, contour listing, and conventional tillage as to their control of fonofos, cyanazine, and atrazine. Other studies have compared: terraced fields with grassed waterways versus contouring for six pesticides (Smith et al. 1978); ridge planting versus conventional tillage for propachlor and diazinon losses (Ritter, 1971); different rotations for pesticide losses (Epstein and Grant, 1968); and no-till versus conservation tillage for pesticide losses (Harvey et al. 1976). More specific information can be found in Sections 4 and 9.

These studies, all excellent by themselves, do not give a complete picture of the interactions between SWCPs and pesticide loss. The studies cover only a small subset of pesticides in current use. Some findings appear to contradict each other, apparently because of climatic and/or soil type differences. Considering the multitude of SWCPs and pesticides, the only practical method of conducting a comprehensive study of SWCP control of pesticide movement is mathematical simulation (Dean and Mulkey, 1978).

The actions of SWCPs on fields have been simulated to determine the effectiveness of SWCPs in reducing pesticide loss in runoff and percolating waters. Pesticides with different characteristics were simulated for this study. In this Section, the results of these simulations are presented.

SIMULATION MODELS

Special Considerations for Pesticide Models

Modeling of pesticide losses differs from the nutrient and sediment modeling presented in the previous two sections. First, nutrient, sediment and pesticide characteristics differ widely. Nutrients and sediment are continuously available for loss in runoff water. Concentrations of these elements in the runoff water, therefore, depend on soil and climatic conditions and not on the time of year per se. Pesticide losses, on the other hand, are greatest when rain occurs soon after application, and decrease thereafter (Ritter et al. 1974; Baker et al. 1978). Thus, the effectiveness of SWCPs can be quite different for pesticides, nutrients and sediment. Second, pesticides usually are degraded before they are leached out of the root zone (Freed and Hague, 1974). Losses in base and interflow are, therefore, of much less concern for pesticides than for nutrients.

Third, pesticides, especially those which are spray applied and not incorporated, have a non-uniform distribution in the soil (Smith et al. 1978). Therefore, there are high and low concentration areas. The pesticide chemistry in the high concentration areas is not well understood, and may be different than when the concentration is uniform throughout the soil (Donigian et al. 1977).

Previous Models

Modeling of pesticide loss in runoff water from agricultural land has been attempted by Crawford et al. (1973), Donigian et al. (1976, 1977), Adams and Kurisu (1976), Frere et al. (1975), Bruce et al. (1975), and McElroy et al. (1976). All but the latter authors have developed continuous, iterative simulation models that are capable of modeling intermediately and strongly adsorbed pesticides. McElroy et al. (1976) have developed a simple loading formula that only applies to strongly adsorbed pesticides.

Two Different Models

In Section 6, the reasons for using more than one simulation model were explained. In summary, models are no better than the assumptions on which they are based. Which assumptions are best is not known at the present time. It is hoped that the use of different models can prevent results from being strongly biased.

Two models were used in this analysis of the pesticide losses. The first was the Agricultural Runoff Management (ARM) Model developed by Hydrocomp, Inc., for the U.S. Environmental Protection Agency. The second was a simulation model developed at Cornell for the specific purpose of evaluating the effectiveness of SWCPs to control pesticide losses.

The ARM Model--

The ARM pesticide model, which is described in detail in Appendix F computes water, soil and pesticide losses in surface runoff and interflow, and downward displacement of the applied pesticides. The hydrologic and sediment portions of the ARM pesticide model are identical to the counterparts in the ARM nutrient model.

The sub-model which handles the movement and degradation of pesticides combines water and sediment flows with a single or multi-valued Freundlich isotherm to determine the amounts of pesticide in solution and in sediment. For modeling of methyl parathion, new algorithms based on the work of Gunther (1977) and Brown (1978) were added by Hydrocomp to represent pesticide storage, decay, and washoff from the crop cover. Attenuation of pesticides through volatilization and degradation is simulated by a first order stepwise degradation function that allows different degradation rates for separate time periods.

ARM Model Calibration--

The ARM model has been calibrated for two watersheds. The first is located near Athens, Georgia, and has the code name of P2. It is one of four plots located on the same type of soil but with different SWCP and cropping patterns. The second calibration watershed, P6, is in Michigan, maintained by the Department of Agricultural Engineering at Michigan State University. Data on pesticide and nutrient loss were obtained under sponsorship of the EPA laboratory in Athens. Characteristics of watersheds are presented in Section 6. Data on pesticide applications are given in Table 7-1.

TABLE 7-1. PESTICIDE APPLICATIONS AT WATERSHEDS P2 IN WATKINSVILLE, GEORGIA, AND P6 IN EAST LANSING, MICHIGAN

Cropping Season	Watershed	Applied Chemical	Application Rate (g/ha)	Date
1973	P2 (Watkinsville)	paraquat	1120	5-11-73
		atrazine	3360	5-11-73
1973	P6 (Michigan)	paraquat	1000	6-09-73
		atrazine	920	11-05-73
1974	P2	paraquat	1120	4-29-74
		atrazine	3360	4-29-74
		paraquat	1120	5-22-74
		atrazine	4480	5-22-74
1974	P6	paraquat	1120	11-08-74
		atrazine	2240	11-08-74
1975	P2	paraquat	1120	5-21-75
		atrazine	1680	5-21-75

Tables 7-2 and 7-3 compare the observed annual water, soil and pesticide losses with those predicted by the model. The simulation results were obtained from Donigian *et al.* (1977). Based on the limited data in Table 7-3 the ARM model does not appear to predict pesticide losses very well for Michigan. The pesticide prediction losses for Georgia (Table 7-2) seem to be more accurate. At both locations, annual runoff was predicted more accurately than were sediment and pesticide losses.

The Cornell Pesticide Model (CPM)--

The Cornell pesticide model consists of five sub-models (Figure 7-1). This approach was chosen so that model components could be changed easily and so that experimental results could be substituted for input data. Model components are discussed below and presented in detail in Appendix B.

The sub-model TEMPMELT takes daily weather records as input data and then computes the rainfall or change in snowpack, if any. The snowmelt

TABLE 7-2. OBSERVED AND ARM-PREDICTED RUNOFF SEDIMENT AND PESTICIDE LOSSES FOR THE P2 WATERSHED (Donigian et al. 1977)

Year		Runoff (cm)	Sediment (kg/ha)	Atrazine (g/ha)	Paraquat (g/ha)
1973	observed	15.9	11,400	64	169
	ARM	15.1	6,900	43[a],28[b]	271
1974	observed	12.1	2,200	6	79
	ARM	13.8	4,400	5[a], 8[b]	242
1975	observed	16.0	5,400	10	183
	ARM	19.3	5,800	8[a], 7[b]	242

[a] based on a multi-valued adsorption/desorption isotherm

[b] based on a single adsorption/desorption isotherm

TABLE 7-3. OBSERVED AND ARM-PREDICTED RUNOFF SEDIMENT AND PESTICIDE LOSSES FOR THE P6 WATERSHED IN EAST LANSING, MICHIGAN

Year		Runoff (cm)	Sediment (kg/ha)	Atrazine (g/ha)	Paraquat (g/ha)
1973	observed	10.9	1,859	9	11
	ARM	12.3	4,163	177[a],196[b]	40
1974	observed	17.1	5,638	52	228
	ARM	20.5	5,687	4[a], 4[b]	286

[a] based on a multi-valued adsorption/desorption isotherm

[b] based on a single adsorption/desorption isotherm

formula is based on that of the U.S. Army Corps of Engineers (1960). Soil temperature at 10 cm depth is predicted as a linear combination of the running average of the previous 20 days' air temperatures (adjusted when there was snow on the ground). The linear combination was then regressed with the observed soil temperature for the years such data were available.

Hydrology--The hydrology sub-model HYDRO assumes, for well-drained, deep soils, that soil water moves readily under gravitational force when its water content is above field capacity. Thus, field capacity is defined here as the moisture content below which water movement is small. It is further assumed that water does not move downward when water content is below field capacity. For shallow soils or where the water table is near the surface, the water cannot move downward but must flow laterally. The number of days that the soil requires to change from saturation to field capacity can be specified.

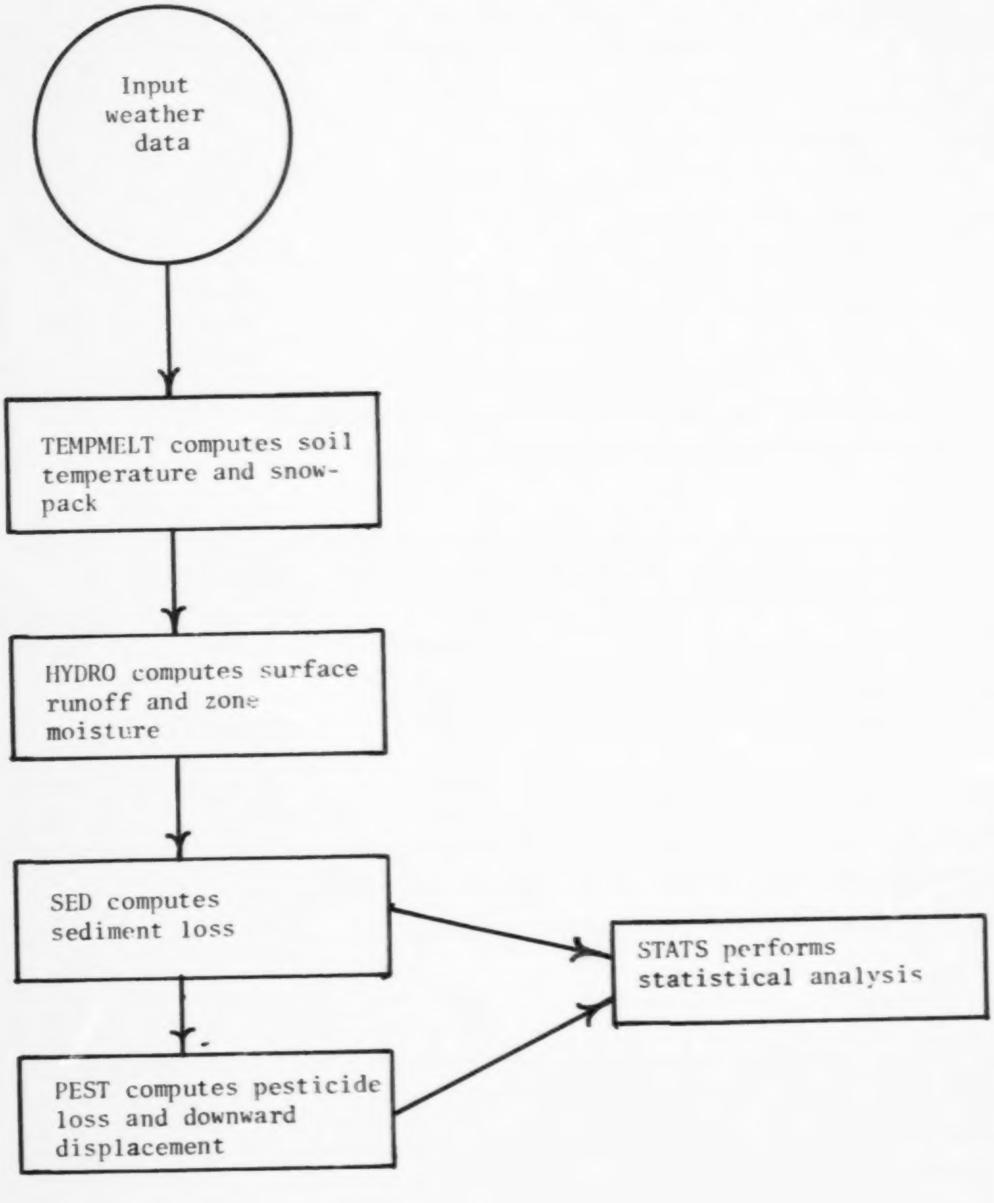
Input
weather
data
TEMPMELT computes soil temperature and snowpack
HYDRO computes surface runoff and zone moisture
SED computes sediment loss
PEST computes pesticide loss and downward displacement
STATS performs statistical analysis

FIGURE 7-1. FLOW CHART FOR CORNELL PESTICIDE MODEL.

The structures of the four-layered root-zone model used is shown in Figure 7-2. Zone 1 is the first 5 cm of topsoil. Zone 2 extends from 5 cm depth to the bottom of the active rooting zone. Zone 3 represents the soil below the active root zone and above the maximum depth of rooting. Zone 4 is the remaining soil depth to the impermeable layer. The above assumptions are essentially equivalent to those used by Woolhiser (Stewart et al. 1976).

Evapotranspiration for growing plants takes place out of Zones 1 and 2 and is assumed to occur at the potential rate if the soil has adequate water (i.e. greater or equal to field capacity). Evapotranspiration is assumed to be zero when the soil is at the wilting point. Actual evaporation rates are, therefore, less than potential as the soil dries. Evaporation during the non-growing season is set at the maximum potential rate when the soil is wet, and decreases with the square root of time thereafter (Black et al. 1969).

The hydrology model has three options to partition rainfall into infiltration and runoff components when the soil temperature is above freezing. The first option is based on the SCS runoff equation (Soil Conservation Service, 1964; Stewart et al. 1976). Boundaries for the antecedent moisture groups are similar to those of the CNS model presented in Section 6. The second option uses infiltration rates based on the location of the wetting front. This method was originally proposed by Green and Ampt (1911) and later modified by Mein and Larson (1971) and Bouwer (1976). This option requires additional input data for rainfall distribution during the storm. Finally for shallow soils, a simple method is used that assumes that all water infiltrates until all pores are filled and hereafter all water runs off. This phenomenon has been observed to occur on soils with fragipans (MacVicar, 1978, Dunne, 1978).

When soil temperature is below freezing, the infiltration model used in the USDAHL74 watershed model (Holtan et al. 1975) was modified such that it could be used in the SCS equation.

This hydrology model is similar in many respects to the simple model of Stewart et al. (1976) and the model described in Section 6. The model does not incorporate the relationships between potential gradient, hydraulic conductivity and water flux in porous media, but instead uses the concept of "field capacity". With the inclusions of the option for shallow soil and allowance for lateral drainage, approximations and assumptions included in the model should not lead to serious errors.

Sediment--The sediment model (SED) uses the Universal Soil Loss Equation with the kinetic energy term computed by either the Onstad and Foster (1975), Williams and Berndt (1977) or Wischmeier and Smith (1965) method. The sediment models are the same as those described in Section 5. The precision and accuracy of the models are also described in Section 5.

Pesticides--The pesticide model consists of three parts. The first part is the prediction of pesticide losses in runoff water. The key assumption of this part is that there exists a certain "mixing" layer at the soil surface where the concentrations of pesticide in the percolating water,

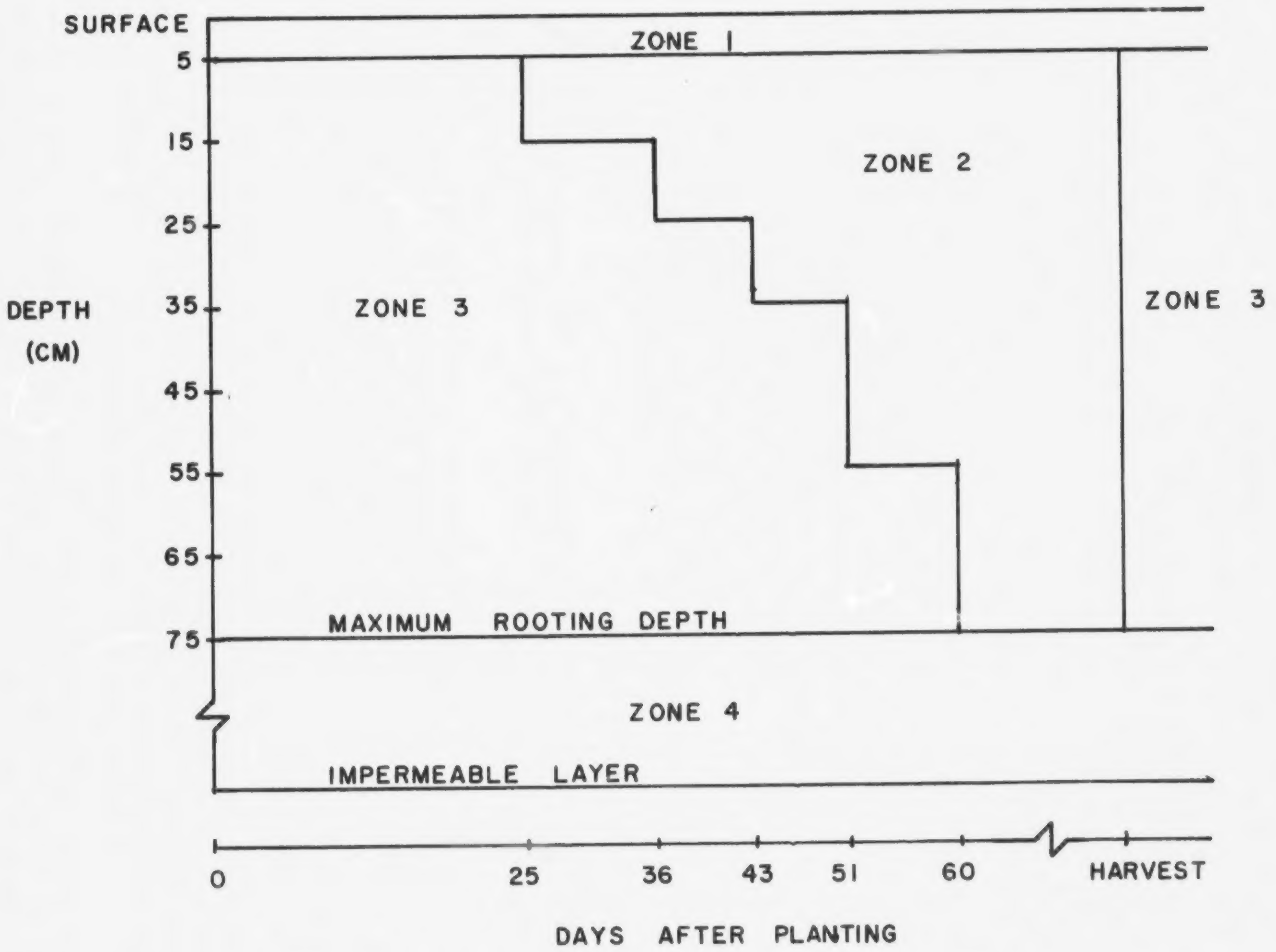
SURFACE
ZONE 1
5
15
25
35
45
55
65
75
DEPTH (CM)
ZONE 2
ZONE 3
ZONE 3
MAXIMUM ROOTING DEPTH
ZONE 4
IMPERMEABLE LAYER
0
25
36
43
51
60
HARVEST
DAYS AFTER PLANTING

FIGURE 7-2. SOIL ZONE BOUNDARIES.

runoff water, and soil water are equal. Theory for predicting the concentration of pesticide in the runoff water is presented elsewhere (Steenhuis and Walter, 1978).

The second part of the model simulates downward displacement of pesticides. It is based on a simple model developed, used and tested successfully by the following authors: Gardner (1965), Bower et al. (1957), and Rao et al. (1976). The displacement of the midpoint of the pesticide band can be found by multiplying a retardation factor by the amount of water passing the band. The value of the retardation factor depends on field capacity of the soil and the adsorption partition coefficient.

The third part of the pesticide sub-model handles the attenuation of pesticide through volatilization and degradation between rainstorms. It is similar in form to that used in the ARM model.

Statistics--The statistical program which was developed as part of this project, summarizes the results produced by the modeling program.

Calibration and Validation--Most of the sub-models of the Cornell pesticide model were developed and tested for accuracy in following modeling studies. The soil temperature, runoff from shallow soils, and pesticide runoff sub-models are exceptions to the above. Predicted and observed soil temperatures at 10 cm depth for the years 1972 and 1973 are shown in Figure 7-3. A good agreement can be noted.

The SCS and Green and Ampt equations used in the hydrology model are in wide use for runoff and infiltration prediction in non-frozen soils. The SCS runoff equation has been tested many times and found to give reasonable results (Stewart et al. 1976; James et al. 1977; Tubbs and Haith, 1977; Soil Conservation Service, 1964). Results presented in Section 6 show that the SCS equation predicted yearly runoff losses somewhat better in New York than in Georgia.

The Green and Ampt equation has recently been reported to predict runoff well (Li et al. 1976; Chu, 1977; Idike et al. 1977). Simulation results of the CPM model in Table 7-4 show that both sub-models varied at times from the observed losses.

For New York, the SCS runoff equation was not used, but instead, the sub-model for shallow soils which assumes that water removal is a function of lateral movement and evapotranspiration. Draining from complete saturation to field capacity was assumed to occur in 20 days. This corresponds approximately to the time that small tributaries of rivers are flowing after the soil is saturated in New York.

The model for shallow soils was verified for Aurora. Observed values for runoff on Aurora plots with a hard pan at 40 cm depth were compared with runoff values predicted by the shallow soil runoff model for the years 1971-1974 (Figure 7-4). A reasonable agreement was obtained for the first two years between predicted and observed values of runoff for well managed plots

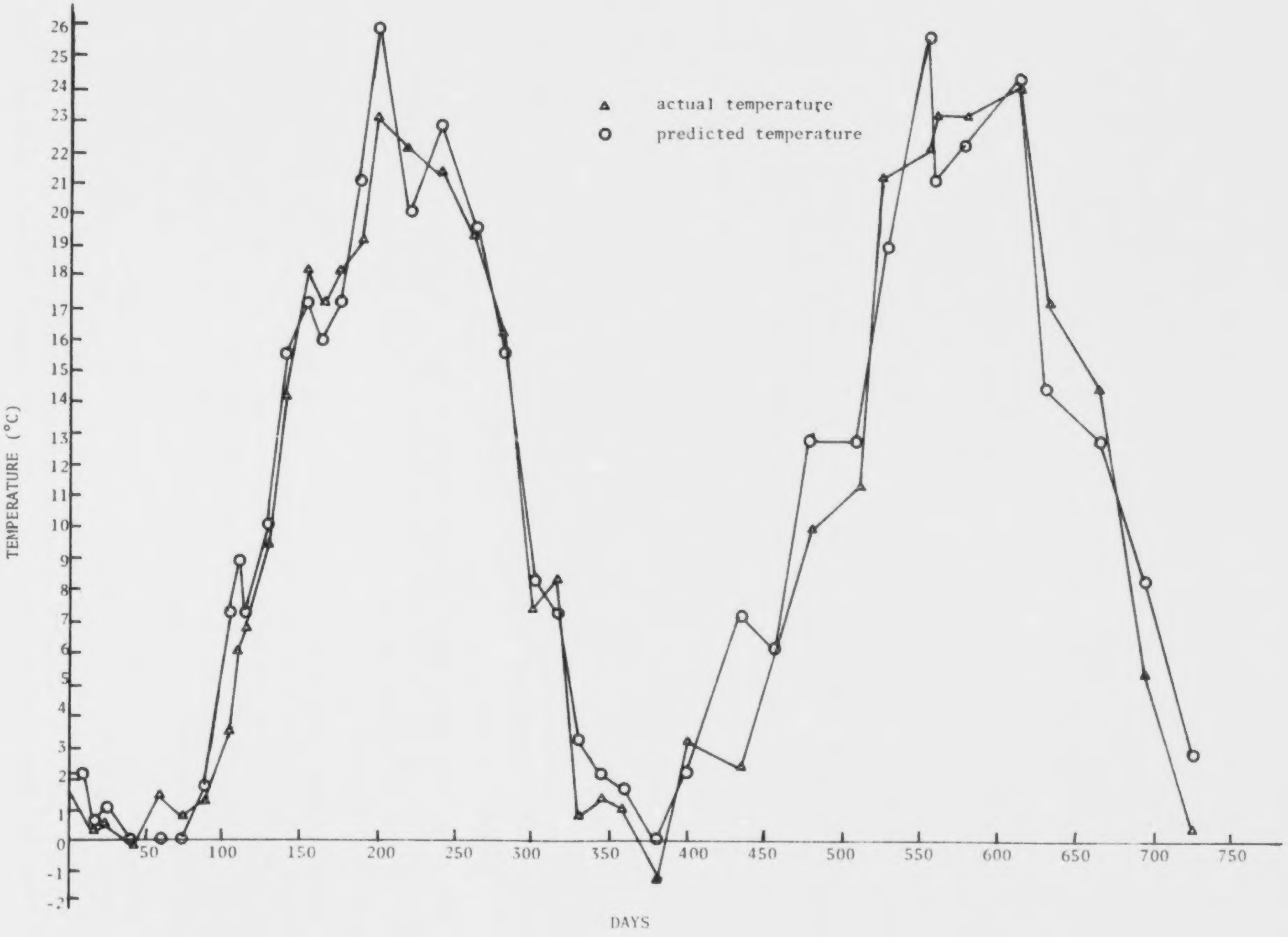

FIGURE 7-3. SOIL TEMPERATURE AT AURORA, N.Y., JAN. 2, 1972 THROUGH Dec. 27, 1973 AT 10 cm.

on shallow soil. The shallow soils model's runoff predictions were as good as those of the SCS curve number equation.

Three soil loss equations were compared in Section 5. It was shown that the Onstad Foster equation (Onstad and Foster, 1975) gave more accurate monthly values for sediment loss than did either the Universal Soil Loss Equation (Wischmeier and Smith et al. 1965) or the Williams equation (Williams and Berndt, 1977). The disadvantage of the Onstad Foster equation is that sediment loss can theoretically occur on days when there is no runoff. Thus, the Onstad Foster equation is preferable only when annual or monthly losses are desired, as in the nutrient simulation of Section 6. The CPM model predicts daily losses of pesticides. Therefore, the CPM predicts soil loss with the Williams model, which restricts its predictions to days when runoff occurs.

The pesticide sub-model has three components, two of which have been validated extensively. The first-order degradation model has been tested by

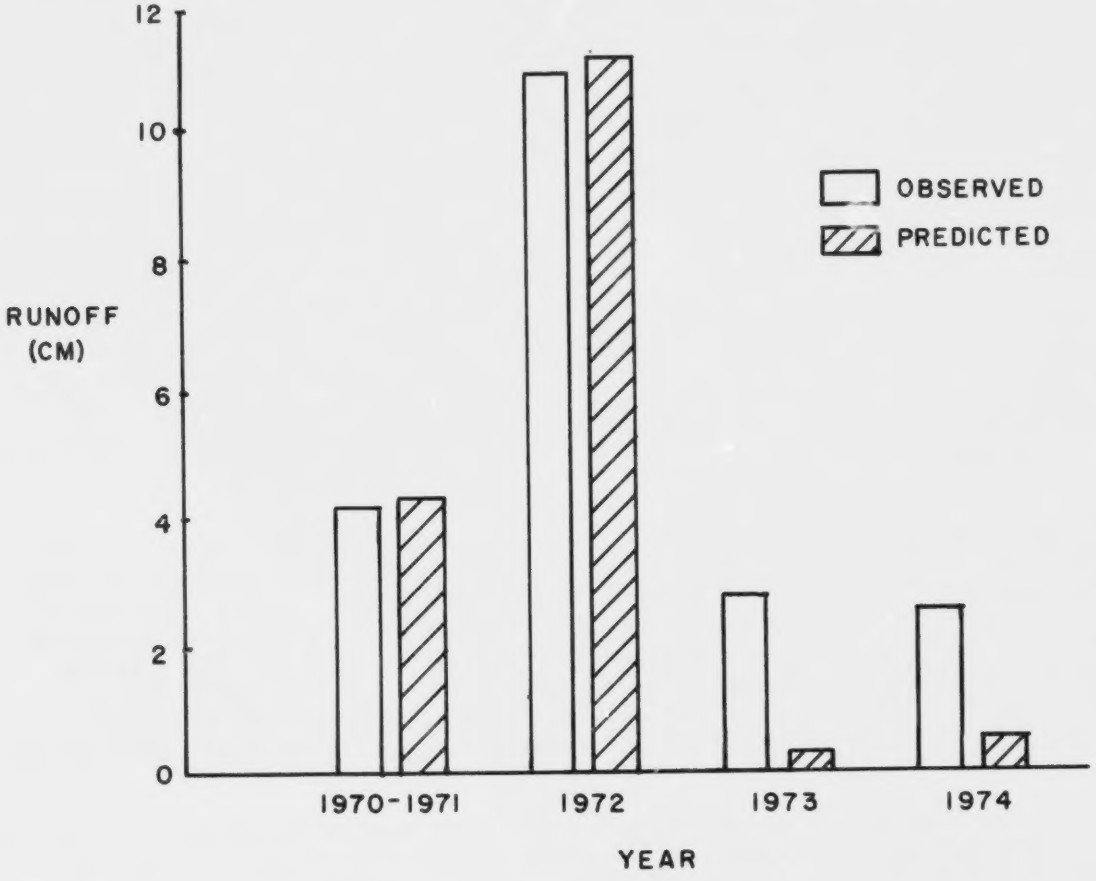

FIGURE 7-4. OBSERVED AND PREDICTED LOSSES FOR AURORA, NEW YORK.

TABLE 7-4. OBSERVED AND PREDICTED RUNOFF LOSSES FOR P2 WATERSHED IN WATKINSVILLE, GEORGIA

	Actual	SCS	Green Ampt	Actual	SCS	Green Ampt	Actual	SCS	Green Ampt
		1973			1974			1975	
	Runoff (cm)								
JAN				0	0.4	0	0.2	1.7	0
FEB				0.4	2.1	0	1.0	2.3	0.3
MARCH				0	0	0	4.8	7.1	2.8
APRIL				0.7	0.1	0.5	1.7	3.2	2.2
MAY	8.0	8.7	5.5	0.8	1.7	1.8	0.9	1.1	0.9
JUNE	3.6	0.1	3.5	4.2	0.9	5.1	4.1	1.7	3.7
JULY	2.0	0.2	1.1	4.6	0.9	5.1	3.2	0.1	1.3
AUG	0	0	0	1.2	0.2	2.7	0.1	0.1	0.9
SEPT	1.1	0.2	1.8	0	0	0			
OCT	0	0	0	0	0	0			
NOV	0	0.1	0	0	0.8	0			
DEC	1.2	4.9	0.9	0.2	1.1	0.1			
Year Total	15.9	14.2	12.8	12.1	8.2	15.3	16.0	17.3	12.1

Donigian et al. (1977) and found to be superior to other methods. The formulas which determine leaching of pesticides were shown to predict the downward movement of substances reasonably well (Rao et al. 1976). The pesticide runoff algorithm is based on parameters that have a physical basis and thus can be measured. An exception is the mixing depth, for which no easy measurable technique exists. A value of 0.9 cm for mixing depth was obtained by Steenhuis and Walter (1978). Their computational procedure incorporated values of alachlor and cyanazine losses measured by Baker et al. (1978).

Data observed by Bailey et al. (1974) were used to validate the pesticide runoff algorithm. Predicted and observed losses were compared by Steenhuis and Walter (1978). A summary of these results is given in Table 7-5. Although observed and predicted losses agreed rather well, the predicted losses were somewhat lower in most cases.

The fonofos concentration in the runoff water measured by Baker et al. (1978) was used to test the sensitivity of the pesticide model to SWCPs. Predicted and observed losses are shown in Figure 7-5 as a percentage of the conventional tillage loss. Trends in pesticide loss under different practices were predicted satisfactorily, as indicated by a correlation coefficient of 73% when observed losses were regressed with predicted losses. However, as shown in Figure 7-5, the observed losses under conventional tillage were lower than for either buffalo till or chisel plowing, while the reverse was true for the predicted losses.

COMPARISON OF THE ARM AND CPM MODELS - 1972-1975

To compare the ARM and CPM predictions of runoff and pesticide loss, both models were run using input data from watershed P2. The ARM model first was calibrated to the watershed. Input data for the ARM simulations are given in Donigian et al. (1977).

The CPM used the Green and Ampt sub-model to compute runoff. Input for the Green and Ampt equations was obtained from the literature: wetting front suction data from Li et al. (1976) and saturated hydraulic conductivity from England (1970). The CPM sediment loss model was based on that of Williams (Williams and Berndt, 1977). Most of the CPM adsorption isotherm for atrazine was made from observed concentrations of the chemical in runoff and sediment (Smith et al. 1978). Outside the range of the observations, the ARM adsorption isotherm was used (Donigian et al. 1977). The paraquat isotherm was the same as in Donigian et al. (1977) adjusted for the difference in mixing depth. Input to the CPM is summarized in Appendix B.

TABLE 7-5. PESTICIDE LOSSES FROM GEORGIA PLOTS, OBSERVED BY BAILEY ET AL. (1974) AND PREDICTED BY THE CPM

		Predicted		Observed	
		total in runoff %	total in sediment %	total in runoff %	total in sediment %
Plot 2	Atrazine	*	*	5	2
	DCBN	*	*	2	T**
Plot 5	Atrazine	8	1	11	2
	DCBN	7	1	8	2
Plot 6	Atrazine	9	1	10	2
	DCBN	7	1	6	3
Plot 7	Atrazine	8	1	7	3
	DCBN	5	2	2	2

* No estimate could be obtained for adsorption partition coefficient

** Trace

Comparison of Results

Simulated runoff losses by ARM and CPM both appear not to be very accurate when compared to observations (Table 7-6). The major shortcoming of both models is that they assume homogeneous fields and overland flow produced equally over the whole watershed.

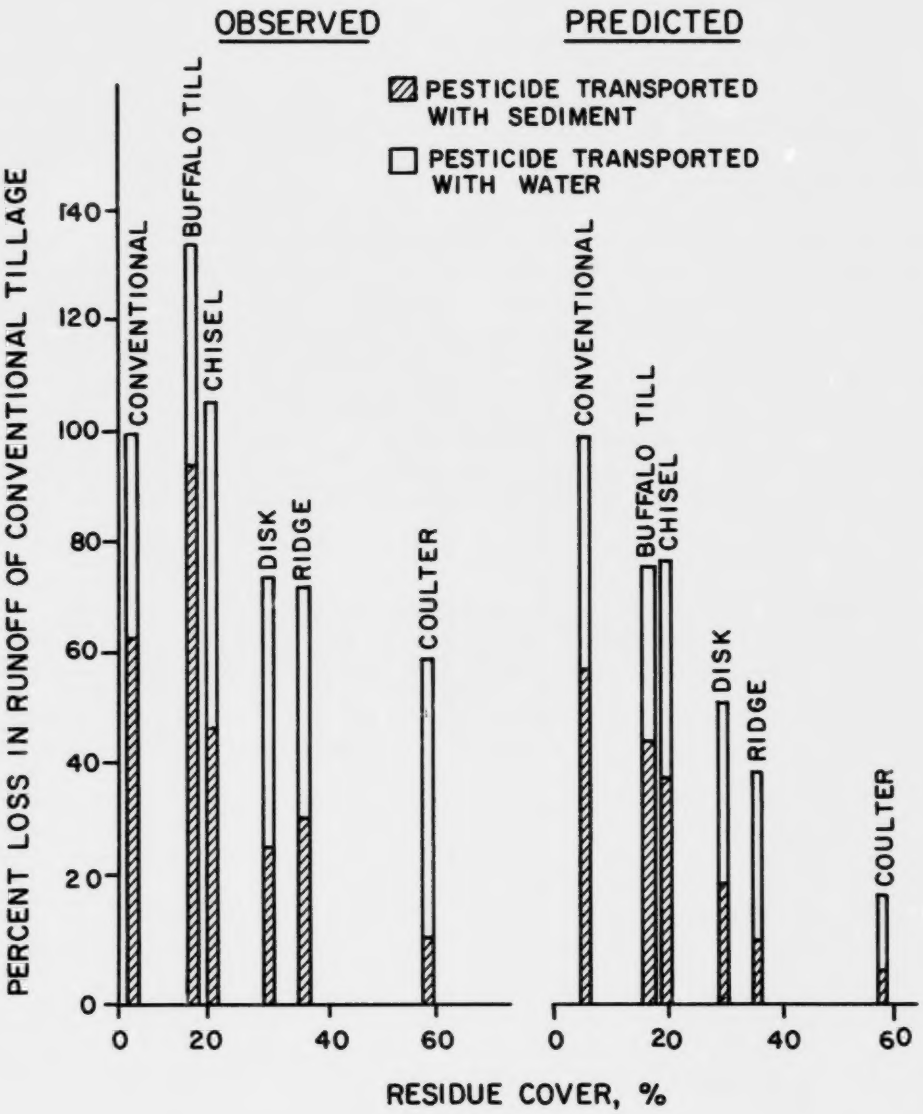

FIGURE 7-5. PESTICIDE LOSS IN RUNOFF AS PERCENT OF LOSS WITH CONVENTIONAL TILLAGE.

TABLE 7-6. ARM AND CPM PREDICTIONS OF RUNOFF FOR THE P2 WATERSHED IN WATKINSVILLE, GEORGIA

	Observed	ARM*	CPM	Observed	ARM	CPM	Observed	ARM	CPM
		1973			1974			1975	
	Runoff (cm)								
JAN				0	0.5	0	0.2	0.3	0
FEB				0.4	0.8	0	1.0	2.0	0.3
MAR				0	0	0	4.8	7.9	2.8
APR				0.7	0.2	0.5	1.7	2.8	2.2
MAY	8.0	7.0	5.5	0.8	1.8	1.8	0.9	2.3	0.9
JUNE	3.6	2.5	3.5	4.2	4.1	5.1	4.1	3.0	3.7
JULY	2.0	0.8	1.1	4.6	4.4	5.1	3.2	1.0	1.3
AUG	0	0	0	1.2	1.9	2.7	0.1	0	0.9
SEPT	1.1	1.8	1.8	0	0	0			
OCT	0	0	0	0	0	0			
NOV	0	0	0	0	0	0			
DEC	1.2	3.0	0.9	0.2	0.1	0.1			
Year Total	15.9	15.1	12.8	12.1	13.8	15.3	16.0	19.3	12.1

*Donigian _et_ _al_. 1977

Simulated atrazine and paraquat losses in runoff water and sediment are compared with observed values in Table 7-7. Annual atrazine losses were predicted better by the ARM than by the CPM model. Paraquat losses were somewhat better simulated by the CPM than by the ARM model.

The CPM is cheaper to run than is the ARM model. A 25-year simulation by the CPM uses less than 1 minute of computer time on an IBM 370/168, at a cost of less than 15 dollars. A 25-year simulation by the ARM cost at least 10 times as much. Furthermore, once soil loss and runoff have been simulated, the effect of pesticide characteristics on pesticide loss can be found within 20 seconds of computer time using the CPM.

The ARM model can predict pesticide losses at any time during the storm. The CPM does not have this capability. The CPM variables have a physical basis. Therefore, the CPM input data are available in the literature whereas the ARM must be calibrated for each watershed. For this reason, the CPM is more appropriate for the studies of regional variations of the effects of SWCPs on pesticide losses.

TABLE 7-7. ARM AND CPM PREDICTIONS OF PESTICIDE LOSSES (GRAMS) FOR THE P2 WATERSHED IN WATKINSVILLE, GEORGIA

	Atrazine Sediment			Atrazine Solution			Paraquat Total		
	Actual	CPM^1	ARM^2	Actual	CPM^1	ARM^2	Actual	CPM^1	ARM^2
1973									
May	10	18	2	65	63	50	188	129	294
June	1	1	0	5	0	4	23	18	29
July	0	0	0	1	0	0	7	35	14
Aug	0	0	0	0	0	0	0	0	0
Sept	0	0	0	0	0	0	0	12	13
Total	11	19	2	71	63	54	218	194	350
1974									
May	1	1	0	6	4	8	23	50	91
June	0	0	0	1	0	0	47	103	164
July	0	0	0	0	0	0	27	98	42
Aug	0	0	0	0	0	0	5	40	15
Sept	0	0	0	0	0	0	0	0	0
Total	1	1	0	7	4	8	102	291	312
1975									
May	1	1	0	1	37*	1	20	37	34
June	1	0	1	10	26	8	191	200	237
July	0	0	0	0	0	0	25	50	27
Aug	0	0	0	0	0	0	0	0	15
Sept	0	0	0	0	0	0	0	13	0
Total	2	1	1	11	63	9	236	300	313

[1] predicted by the Cornell Pesticide Model

[2] predicted by the Agricultural Runoff Model by Hydrocomp based on non-single valued adsorption isotherm

*too high because of over-prediction of runoff by CPM

EVALUATION OF THE EFFECTS OF SWCPs ON PESTICIDE LOSSES

The ARM and CPM were used to evaluate the effects of management practices on pesticide losses. Atrazine, paraquat, and methyl parathion losses were modeled using the ARM for a 10-year period with the same management practices and locations as in the nutrient simulations described in Section 6 and Appendix F. Four groups of pesticides were modeled using the CPM for 25-year periods at locations in Georgia, Iowa and New York. Two management practices were evaluated: a) straight row planted corn, and b) corn on terraces combined with contour planting.

The ARM Comparisons

The ARM model, which has been calibrated and tested on the P2 (Georgia) and P6 (Michigan) watersheds, was used to model three pesticides for the 10-year period 1966 through 1975 at these two locations. The management practices were: a) base condition (straight row corn, disking in spring and no plant residues during the winter), b) contours (rows perpendicular to the land slope), c) terraces combined with contours, and d) no tillage (crop residue left on field after harvest, and no tillage prior to spring planting).

The ARM model is not directly sensitive to SWCPs. Therefore, input parameter changes were based on either the SCS runoff equation, the Universal Soil Loss Equation or Hydrocomp's experience of the model's behavior on other watersheds. The estimation procedure for these input parameters is presented in Appendix F.

Atrazine, paraquat, and methyl parathion were chosen because they are representative of broad groupings of pesticides. Atrazine, as modeled by Hydrocomp, is an intermediately adsorbed pesticide with a relatively short half-life (15 days). Paraquat is a strongly adsorbed pesticide with high persistence (half-life of 360 days). Methyl parathion, which was applied to the canopy six times per year, is intermediately adsorbed and has an intermediate half-life (about 40 days). Application days and rates are given in Table 7-8.

TABLE 7-8. PESTICIDE APPLICATIONS ON WATERSHEDS P2 and P6 FOR ARM MODEL SIMULATIONS

	Atrazine		Paraquat		Methyl Parathion	
	Date Applied	Rate (kg/ha)	Date Applied	Rate (kg/ha)	Date Applied	Rate (kg/ha)
P2	May 9	2.9	May 9	2.5	June 15	1.1
					June 29	1.1
					July 13	1.1
					July 27	1.1
					August 10	1.1
					August 24	1.1
P6	May 20	2.8	May 20	1.5		

ARM Simulation Results and Discussions

Table 7-9 summarizes the mean annual runoff and pesticide losses (in sediment and in solution) predicted by Hydrocomp, Inc. for each of the management practices on the P2 and P6 watersheds. Since application rates for the two watersheds were not equal, results are also given in percent of total pesticide applied. The results show that the annual runoff pesticide losses in Georgia and Michigan were decreased when any of the three SWCPs were incorporated. Runoff was reduced by 6 to 11% for contours and by 25 to 36% for terraces (Table 7-10). There were lesser reductions (6%) for no tillage. Simulated sediment losses for the management practices reflect the reductions in surface runoff. For both locations, the percent change in sediment loss from base condition for contouring was 21 to 37%, for terraces 32 to 55%; and for no tillage 48 to 71%. These changes are comparable to those reported by others as noted in the following paragraphs.

Annual runoff losses for corn planted on the contour in experiments in Illinois, Missouri, New Jersey and Ohio (Stallings, 1945) were approximately

TABLE 7-9. P2 AND P6 MEAN ANNUAL RUNOFF AND POLLUTANT LOSSES UNDER DIFFERENT MANAGEMENT PRACTICES MODELED BY THE ARM MODEL FOR A 10-YEAR PERIOD

	Base Condition		Minimum Tillage		Contours		Terraces and Contours	
	P2	P6	P2	P6	P2	P6	P2	P6
Runoff*(cm)	19.9	19.1	18.6	18.3	17.7	17.9	12.8	14.4
Sediment Loss (mt/ha)	6.3	3.5	3.3	10.	4.0	2.8	2.8	2.4
PESTICIDE LOSSES								
Agrazine								
in solution (g/ha)	53.1	77.4	46.9	75.7	40.6	61.7	18.8	28.9
as % of applied	1.8	2.8	1.6	2.7	1.4	2.2	0.6	1.0
on sediment (g/ha)	0.4	1.8	0.3	1.1	0.3	1.3	0.2	0.9
as % of applied	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Methyl Parathion								
in solution (g/ha)	283		246		241		145	
as % of applied	4.3		4.2		3.6		2.2	
on sediment (g/ha)	23		18		15		10	
as % of applied	0.4		0.3		0.2		0.2	
Paraquat**								
total (g/ha)	341	115	188	36	225	93	162	81
as % of applied	13.6	7.7	7.5	2.4	9.0	6.2	6.5	5.4

* Overland flow plus interflow

**Paraquat losses in the solution phase are assumed negligible

TABLE 7-10. P2 AND P6 MEAN ANNUAL RUNOFF AND POLLUTANT LOSSES UNDER DIFFERENT SWCPs AS PERCENT CHANGE FROM BASE CONDITION LOSSES MODELED BY ARM FOR A 10-YEAR PERIOD

	Minimum Tillage		Contours		Terraces and Contours	
	percent					
	P2	P6	P2	P6	P2	P6
Runoff*	- 6	- 4	-11	- 6	-36	-25
Sediment loss	-48	-71	-37	-21	-55	-32
Pesticide Losses						
Atrazine						
in solution	-12	- 2	-24	-20	-65	-63
in sediment	-17	-40	-32	-31	-61	-53
combined	-12	- 3	-24	-20	-64	-62
Methyl Parathion						
in solution	-13		-14		-49	
in sediment	-34		-45		-66	
combined	-15		-17		-50	
Paraquat	-45	-68	-34	-19	-52	-29

* Overland flow plus interflow

30% lower than losses for corn planted in straight row. Contours at these same locations reduced soil loss by about 50%. More recent studies (Stewart et al. 1976) showed reductions similar to those quoted by Stallings (1945).

The effect of terraces on runoff and soil loss depends mainly on the type of terrace and the permeability of the soil (Baver et al. 1972). Studies quoted by Stallings (1945) and in Stewart et al. (1976) showed that terraces were more effective in reducing soil loss than runoff. Data presented by Stallings (1945) show that terraces in most cases reduced runoff 20 to 50% as compared to runoff from unterraced fields in Arkansas, Iowa, Missouri, Oklahoma and Texas. However, runoff was approximately 20% greater on the terraced than on the unterraced fields in Ohio and Wisconsin. Soil loss for all locations was reduced, with reductions ranging from 35% on a 3% slope in Iowa to 98% in Missouri on an 11% slope.

The reductions in runoff and soil loss predicted by the ARM simulation for terraces and contours, although somewhat small, agree reasonably well with the above observations. However, reductions in runoff and sediment loss under no tillage seem to be underestimated. Jamison et al. (1968) and Free and Bay (1969) reported reduction of less than 15% for runoff. However, their results were for a soil with a fragipan, and are not applicable to the soils of Georgia and Michigan. Most other studies, such as those by Harrold et al. (1970), and Shanholtz and Lillard (1968), indicate reductions in runoff surpassing 40%. No-tillage reductions in sediment loss are most times greater than 90% (Harrold and Edwards, 1972; Harrold et al. 1970).

In the ARM model, pesticide losses are directly related to runoff and sediment losses. Therefore, if runoff and sediment losses are not predicted correctly, neither will be the pesticide losses. Predictions of pesticide losses under no-tillage should, therefore, be looked at critically. Simulated average annual overland losses of methyl parathion for the base condition are 4.6% of total amount applied. The few experimental studies currently available indicate that runoff and sediment losses of methyl parathion are minor and amount to a maximum of 0.1% of the applied material (Sanborn et al. 1976, Pesticide manual, 1971; Von Rumker, 1975). The ARM model seems, based these few studies, to overpredict methyl parathion losses. Cause for this might be the difference in drift and volatilization between actual and simulated conditions. Drift and volatilization losses are high in the field (Hague and Freed, 1974, Quinby et al. 1958) but neglected in the ARM model. Another cause for the higher simulated losses is that the degradation rate in ARM is lower than observed under field conditions. Lichtenstein and Schulz (1964) found that only 8% of the 5 pounds per acre of methyl parathion remained 15 days after application. King and McCarty (1968) estimating the theoretical half-life have shown similar results. ARM model assumed half-life for methyl parathion degradation of 49 days.

Methyl parathion and atrazine are mainly lost as solutes transported in the runoff water (Table 7-9). Therefore, total reductions of pesticide losses are more related to the reductions in runoff water than in the sediment (Table 7-10).

ARM simulation results show that contours combined with terraces in Georgia and Michigan are the most effective in reducing the losses of intermediately adsorbed pesticides such as atrazine or methyl parathion. Contours alone are not as effective in reducing losses of these pesticides as terraces combined with contours are.

Paraquat is strongly and irreversibly adsorbed to soil particles and, therefore, paraquat loss corresponds to sediment loss. Small variations are caused by the quantity of fines transported in the runoff--the fines are the relatively strongest adsorbing particles. Therefore, minimum tillage, which has its greatest effect on sediment transport, is the most effective practice for reducing paraquat losses.

Three principal conclusions can be drawn from the ARM simulations: 1) The changes in loss caused by SWCPs are simulated for the strongly adsorbed pesticides such as paraquat and the intermediately adsorbed pesticides such as methyl parathion and atrazine; 2) Absolute quantities of paraquat are more reduced than either atrazine or methyl parathion; 3) Average annual pesticide losses form a small percentage of the total amount of applied pesticide.

The CPM Simulation's 25-Year Runs

In this section, the effect of two SWCPs and four groups of pesticides on soil, water and pesticide losses are evaluated at three locations over a 25-year period by the CPM model. The two SWCPs, terraces and straight row, represented the good and poor extremes in terms of preventing pollutant losses in overland flow. The pesticide classification was chosen such that most pesticides currently on the market were represented.

The CPM simulations were made for the same locations and years (1952-1976) as for the CNS simulation in Section 6 and the erosion studies in Section 5. Relevant input parameters were kept approximately equal to those of the two preceding sections with the exception of slope, which was kept constant at 2% to facilitate the evaluation of climatic effect of pesticide loss. Soil and crop characteristics and pesticide application dates are summarized in Table 7-11.

Local climatic data were input for Watkinsville, Georgia; Ames, Iowa; and Aurora, New York. Runoff and infiltration simulations in Georgia and Iowa were based on the SCS curve number method. For New York, the subprogram for prediction of runoff on shallow soils with a hard pan was used. Erosion was modeled with the Williams model (Williams and Berndt, 1977) for reasons given before.

Simulations were done for four groups of pesticides with similar properties:

Group A	weakly adsorbed and easily degradable
Group B	intermediately adsorbed and easily degradable
Group C	intermediately adsorbed and intermediately persistent
Group D	strongly adsorbed and very persistent

Although many classifications are possible, it was thought that the above classification was the most suitable for evaluating pesticide loss in runoff. To demonstrate grouping according to persistence and adsorption strength, selected pesticides are classified in Table 7-12.

In the CPM simulation studies, all pesticides were applied on May 10 at a rate of 3 kg active ingredient per ha. In a seperate simulation, the Group B pesticides were also applied on June 10 at 3 kg per ha.

Complete simulations were made for straight-row-planted corn in poor hydrological condition and for corn planted on the contour combined with terraces and a good hydrological condition. These two cases are the worst and best, respectively, in terms of preventing loss.

Water and Sediment Losses--

Simulation results are presented as average annual values and as monthly frequencies. The average annual value is the cumulative 25-year value divided by 25. Monthly frequencies are the 25-year totals of the number of times an event occurs in each month of the year.

TABLE 7-11. SOIL AND CROP CHARACTERISTICS FOR STUDY LOCATIONS

	Georgia	Iowa	New York
I. Soil Characteristics	Cecil Sandy Loam	Tama Silt Loam	Kendaia Silt Loam
Hydrologic group	B	B	n.a.*
Erodibility	0.30	0.30	0.30
Area (ha)	1.3	1.3	1.3
Slope (%)	2	2	2
Slope length (m)	180	180	180
Bulk density (g/cm^3)	1.65	1.22	1.31
Field capacity (cm^3 · cm^{-3})	0.28	0.44	0.33
Saturation (cm^3 · cm^{-3})	0.39	0.55	0.50
Wilting point (cm^3 · cm^{-3})	0.14	0.22	0.16
Depth to impermeable layer (cm)	n.a.	n.a.	40
II. Crop and Field Data			
crop	continuous corn	continuous corn	continuous corn
plowing time	spring plow	spring plow	spring plow
pesticide applications	May 10**	May 10**	May 10**
emergence data	June 1	May 15	June 1
harvest date	Nov. 1	Nov. 10	Nov. 1
CN fallow season (AMC = 2)	86	86	n.a.
CN growing season (AMC = 2)	81	81	n.a.
maximum rooting depth (cm)	75	135	40

* not applicable

** June 10 pesticide application also modeled for group B pesticides

Precipitation Analysis--

The average annual precipitation at the Georgia, Iowa, and New York locations for the 25-year simulation period (1952-1976) was 129, 80 and 88 cm, respectively. The frequency analysis for rainfall in these locations is presented in Table 7-13.

Climates in Iowa and Georgia are characterized by heavy thunderstorms in late spring. There are relatively few heavy rainfalls in the summer. Highest rainfall intensities occur, thus, at the time that most herbicides are applied and the soil is not protected by plant cover. The climate in New York is characterized by many days of precipitation, but the rains are usually of low intensity. Only in summer do heavy rainfalls occur. As opposed to Georgia, winter precipitation in Iowa and New York is often in the form of snow. Thus, Georgia receives the most rain and the most severe storms.

TABLE 7-12. CLASSIFICATION OF SELECTED PESTICIDES[a]

Group A	Group B	Group C	Group D
weakly adsorbed and easily degradable	intermediately adsorbed and easily degradable	intermediately adsorbed and intermediately degradable	strongly adsorbed and very persistent
Chloramben[b]	Atrazine[c,d] Carbaryl[e] Malathion[e] Methyl Parathion[g] Phorate[h] Propachlor[i] Trifluralin[c,e]	Atrazine[c,d] Trifluralin[c,e]	Paraquat[d] Toxaphene[f]

[a] adsorption characteristics from Stewart et al. (1975)
persistent data from authors as indicated

[b] Sanborn et al. (1977)

[c] degradation rate dependent on soil type and weather conditions

[d] Leonard et al. (1976)

[e] von Rumker et al. (1974)

[f] Guyer et al. (1971)

[g] Lichtenstein and Schulz (1964)

[h] Miller and Berg (1969)

[i] Hilton et al. (1974)

Runoff Analysis--

Of the three locations, Georgia had the highest annual runoff, in absolute amount and also as a percentage of the rainfall. New York had the lowest annual runoff (Table 7-14). But more important, Georgia also had the greatest number of runoff events in excess of 5 cm -- one about every four years (Table 7-15). This occurred whether the corn was planted on straight row or on terraces. Iowa had three runoff events in a 25-year period in excess of 5 cm -- one for corn planted straight row, and two when terraces were installed. New York had only one runoff event greater than 5 cm (and this was due to a hurricane). These large storms carry a more-than-proportional share of the total pollutant load. Thus, although New York had the greatest number of rain days, Georgia had the greatest number of days that runoff occurred. In addition to daily rainfall, it should be noted that the soil itself is an important factor in determining runoff magnitude. In Aurora, New York, soils are highly permeable and runoff does not occur until the profile is completely saturated. In Georgia and Iowa, test plots have a lower soil permeability, such that runoff can occur even if the soil is not saturated, and increases as moisture content increases.

TABLE 7-13. FREQUENCY ANALYSIS FOR RAINFALL IN GEORGIA, IOWA, AND NEW YORK

	Probability of a dry day			Probability of daily rainfall occurring in excess of											
				0 cm/day			1 cm/day			5 cm/day			10 cm/day		
	Georgia	Iowa	N.Y.	Georgia	Iowa	N.Y.	Georgia	Iowa	N.Y.	Georgia	Iowa	N.Y.	Georgia	Iowa	N.Y.
	percent														
January	63.6	81.0	54.0	36.4	19.0	46.0	12.7	1.5	3.6	0.4	0.0	0.0	0.0	0.0	0.0
February	64.7	70.3	53.1	35.3	21.7	46.1	14.5	1.4	4.0	0.5	0.1	0.2	0.0	0.0	0.0
March	64.9	71.7	54.7	35.1	28.3	45.3	14.4	4.5	5.6	1.7	0.1	0.0	0.0	0.0	0.0
April	71.3	67.2	53.6	28.7	32.8	46.4	11.1	9.6	7.4	0.8	0.3	0.0	0.0	0.0	0.0
May	70.8	64.0	52.9	29.2	36.0	47.1	12.3	11.8	9.8	1.2	0.6	0.0	0.2	0.1	0.0
June	69.2	65.4	61.0	30.8	34.6	39.0	10.3	12.7	8.1	1.2	1.3	0.4	0.2	0.1	0.0
July	63.7	70.1	64.7	36.3	29.9	35.3	11.5	8.4	8.3	0.6	0.5	0.2	0.2	0.0	0.1
August	72.2	73.6	63.7	37.7	27.4	36.3	9.7	7.8	8.4	0.9	0.7	0.4	0.0	0.1	0.0
September	76.9	70.5	63.8	33.1	29.5	36.2	7.8	8.5	7.0	0.6	0.3	0.2	0.1	0.1	0.0
October	80.7	77.1	59.6	19.3	22.9	40.4	6.8	5.8	7.5	0.6	0.2	0.3	0.1	0.0	0.0
November	73.4	80.1	54.4	26.6	19.9	46.6	10.2	3.4	6.9	0.4	0.0	0.0	0.0	0.0	0.0
December	66.8	81.3	51.0	33.2	17.7	49.0	11.2	1.8	6.4	0.8	0.0	0.2	0.0	0.0	0.0

TABLE 7-14. MEAN ANNUAL RUNOFF AND SEDIMENT LOSSES FOR TWO MANAGEMENT PRACTICES

	Runoff			Sediment	
	cm	% rainfall	% change	mean MT/ha	% change
Georgia					
straight row	17.1	13	--	18.0	--
terraces	15.8	12	- 7	3.0	-83
Iowa					
straight row	6.2	8	--	6.2	--
terraces	4.8	6	-22	0.6	-90
New York					
straight row	2.3	3	--	0.8	--
terraces	2.3	3	0	0.2	-75

Local climatic conditions had a much greater effect on magnitude and time of runoff events than did management practices (Tables 7-15 and 7-16). On the shallow fragipan soil in New York, soil management practices did not reduce runoff at all, while reductions in Iowa and Georgia were small (7-22% reductions when terraces were installed). The differences in runoff due to location was much more significant -- there was 87% less runoff in New York than in Georgia.

Soil Loss Analysis--

Average annual soil loss followed the same pattern as the average annual runoff losses (Table 7-14). However, SWCPs do have the potential to correct for unfavorable local climatic conditions and control soil loss. With terraces, not only was average soil loss reduced by at least 75% for all locations, but soil loss did not exceed 5 tons per hectare for any storm (Table 7-16). The frequencies of occurrences of soil loss of all magnitudes were less with terraces.

Runoff and soil loss for other management practices are presented in Sections 5 and 6. All the SWCPs controlled soil loss better than they controlled runoff. This is an important conclusion for explaining the interaction of pesticide characteristics and SWCPs in the following section.

Pesticide Losses--

Mean annual pesticide losses are given in Table 7-17 for the four groups of pesticides applied on May 10 of each year, and in Table 7-18 for the pesticides in Group B applied on May 10 and June 10. For the weakly and intermediately

TABLE 7-15. MONTHLY RUNOFF FREQUENCY ANALYSIS

	Number of runoff events in a 25 year period which are						
	0 - 1 cm/day					1 - 5 cm/day	
	Georgia		Iowa		New York[1]	Georgia	
	straight row	terraces	straight row	terraces	straight row and terraces	straight row	terraces
January	82	82	19	19	2	22	22
February	98	98	31	31	6	14	14
March	74	74	52	52	7	22	22
April	42	42	56	56	2	15	15
May	47	46	29	10	-	9	8
June	24	13	37	14	2	2	-
July	20	7	11	5	1	1	1
August	17	7	15	6	3	-	-
September	17	5	15	2	1	3	1
October	9	5	5	2	2	3	1
November	53	53	14	14	0	9	9
December	73	73	15	15	5	17	17

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[1] In New York, runoff is the same for both terraces and straight row.

TABLE 7-15. MONTHLY RUNOFF FREQUENCY ANALYSIS (con't)

Number of runoff events in a 25 year period which are							
1 - 5 cm/day			Greater than 5 cm/day				
Iowa		New York[1]	Georgia		Iowa		New York[1]
straight row	terraces	straight row and terraces	straight row	terraces	straight row	terraces	straight row and terraces
2	2	1	1	1	-	-	-
2	2	5	-	-	1	1	-
5	5	2	3	3	-	-	-
13	13	-	-	-	-	-	-
6	3	-	1	1	1	1	-
3	1	2	2	2	-	-	-
1	-	-	-	-	-	-	1
1	2	-	-	-	1	-	-
1	1	1	-	-	-	-	-
-	-	1	-	-	-	-	1
1	1	-	-	-	-	-	-
2	2	4	-	-	-	-	-

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[1] In New York, runoff is the same for both terraces and straight row.

TABLE 7-16. MONTHLY SOIL LOSS FREQUENCY ANALYSIS

		Number of days that soil loss is 0.1 - 1 MT/ha day			1 - 5 MT/ha day			5 - 10 MT/ha day			Greater than 10 MT/ha day		
		Georgia	Iowa	New York	Georgia	Iowa	New York	Georgia	Iowa	New York	Georgia	Iowa	New York
January	straight row	57	--	--	4	--	--	--	--	--	--	--	--
	terraces	33	--	--	--	--	--	--	--	--	--	--	--
February	straight row	46	--	--	5	--	--	--	--	--	--	--	--
	terraces	25	--	--	--	--	--	--	--	--	--	--	--
March	straight row	45	--	--	18	--	--	--	--	--	--	--	--
	terraces	44	--	--	--	--	--	--	--	--	--	--	--
April	straight row	32	34	--	9	4	--	1	--	--	--	--	--
	terraces	26	--	--	1	--	--	--	--	--	--	--	--
May	straight row	22	13	--	27	13	--	6	2	--	1	1	--
	terraces	39	22	--	7	2	--	--	--	--	--	--	--
June	straight row	16	26	--	10	11	1	--	--	--	2	--	--
	terraces	6	9	3	2	--	--	--	--	--	--	--	--
July	straight row	12	8	1	8	3	--	1	--	--	1	--	--
	terraces	4	4	1	--	--	--	--	--	--	--	--	--
August	straight row	12	14	1	4	3	--	--	--	--	--	--	--
	terraces	4	4	1	--	--	--	--	--	--	--	--	--
September	straight row	13	10	--	4	1	2	--	--	--	--	--	--
	terraces	5	1	2	--	--	--	--	--	--	--	--	--
October	straight row	6	4	--	4	--	2	--	--	--	--	--	--
	terraces	4	--	2	--	--	--	--	--	--	--	--	--
November	straight row	32	7	--	6	--	--	--	--	--	--	--	--
	terraces	23	4	--	--	--	--	--	--	--	--	--	--
December	straight row	42	--	--	--	--	--	--	--	--	--	--	--
	terraces	31	--	--	--	--	--	--	--	--	--	--	--

TABLE 7-17. AVERAGE ANNUAL PESTICIDE LOSSES APPLIED EACH YEAR ON MAY 10

	straight row			straight row		
	Georgia	Iowa	New York	Georgia	Iowa	New York
Group A (low k*, $t_{1/2}$ **=15 days)						
in solution (g/ha)	11.6	19.9	T***	11.0	6.3	T
as percent of applied	0.4	0.7	T	0.4	0.2	T
in sediment (g/ha)	4.1	2.6	T	0.7	0.2	T
as percent of applied	0.1	0.1	T	T	T	T
Group B° (intermediate k, $t_{1/2}$ = 15 days)						
in solution (g/ha)	26.3	40.0	T	26.0	16.0	T
as percent of applied	0.9	1.3	T	0.9	0.5	T
in sediment (g/ha)	9.0	4.9	T	0.4	0.4	T
as percent of applied	0.3	0.2	T	T	T	T
Group C° (intermediate k, $t_{1/2}$ - 100 days)						
in solution (g/ha)	45.4	54.5	0.7	42.4	19.7	0.7
as percent of applied	1.5	1.8	T	1.4	0.7	T
in solution (g/ha)	10.8	7.8	0.5	0.6	0.6	0.1
as percent of applied	0.4	0.3	T	T	T	T
Group D°° (high k, $t_{1/2}$ = 360 days)						
total (g/ha)	536.2	313.6	48.8	139.6	64.4	26.7
as percent of applied	17.9	10.4	1.6	4.7	2.2	0.9

* k = adsorption partition coefficient

**t1/2 = half-life

*** = Trace, less than 0.05

° adsorption isotherm equal to atrazine in appendix B

°° adsorption isotherm equal to paraquat in appendix B

TABLE 7-18. AVERAGE ANNUAL PESTICIDE LOSSES (IN GROUP B) FOR TWO DAYS OF APPLICATION

	straight row			terraces		
	Georgia	Iowa	New York	Georgia	Iowa	New York
Applied May 10						
in solution (g/ha)	26.3	40.0	T*	26.0	16.0	T
as % of applied	0.9	1.3	T	0.9	0.5	T
in sediment (g/ha)	9.0	4.9	T	0.4	0.4	T
as % of applied	0.3	0.2	T	T	T	T
Applied June 10						
in solution (g/ha)	3.8	8.6	1.0	0.5	1.6	1.0
as % of applied	0.1	0.3	T	T	0.1	T
in sediment (g/ha)	2.0	1.6	0.4	T	T	T
as % of applied	0.1	0.1	T	T	T	T

* T = trace, less than 0.05

adsorbed pesticides, most loss occurred in the solution phase. Pesticide losses for both Iowa and Georgia were significantly higher than for the shallow fragipan soil in New York. Pesticide loss from New York soils with the exception of the most strongly adsorbed persistent pesticides, does not appear to be an environmental problem. Pesticide losses for the June 10 application are much lower than for the May 10 application for both Georgia and Iowa; conversely, a slight increase was noted for New York.

For the three locations, Tables 7-19, 7-20, and 7-21 show the number of days with a pesticide loss in a 25-year period. For example in Georgia for corn planted straight row, there were for Group A, 40 days in May over a 25-year period that pesticides could be detected in the overland flow. Of these 40 days, 27 days had an amount of pesticide lost below 0.3% of the total amount applied and 13 days an amount between 0.3 and 4% of that applied.

For Iowa and Georgia, the May 10 application is at the beginning of the period of heavy thunderstorms and consequent high runoff events (see previous section). The June 10 application falls towards the end of this period. Tables 7-19 and 7-20 clearly show this. The number of runoff events with measurable pesticide losses is reduced significantly for the June 10 application (of pesticides in Group B) when compared with the May 10 application of the same pesticide.

For New York, the large runoff events occur in summer when heavy thunderstorms occur (Table 7-15). Again, the closer the date of pesticide application to the period of intense rainfall, the greater the chance of pesticide loss. This is illustrated in Table 7-21.

Adsorption partition coefficient and half-life greatly affected pesticide loss. The higher the half-life and adsorption partition coefficient, the greater was the pesticide loss. For straight row corn, as much as 18% of the applied Group D pesticide was lost in Georgia. The longer the half-life and the stronger the adsorption of the pesticide, the greater were the losses which occured in the month of application.

In Georgia, terraces were not very effective in reducing the quantity of pesticides transported in the runoff water if the date of application was May 10. In Iowa, as opposed to Georgia, terraces are much more effective in reducing the dissolved component of the pesticide loss. The main reason for this difference is that the soil has a lower moisture content in May in Iowa than in Georgia. Low evaporation and relatively large rainfall quantities in the winter in Georgia cause this high moisture content. The water infiltrated before runoff occurs (or initial abstraction) computed with the SCS curve number is much greater for dry conditions than for wet conditions. The more water that infiltrates before runoff occurs, the more pesticide is leached out of the surface zone and the less is lost in surface water. For all locations, terraces greatly reduce the portion of pesticide transported with the sediment. The decrease in loss of strongly adsorbed pesticides is almost as great as the decrease of sediment (Table 7-16).

Pesticides of Groups A, B, and C are mostly transported in the water phase. Therefore, the effect of SWCPs on runoff is the determining factor

TABLE 7-19. FREQUENCY OF NUMBER OF PESTICIDE LOSSES IN A 25-YEAR PERIOD UNDER TWO MANAGEMENT PRACTICES IN GEORGIA

		Group A appl. May 10			Group B appl. May 10			Group B appl. June 10			Group C appl. May 10			Group D appl. May 10		
		0 to 0.3 percent of applied lost per day	0.3 to 4	4 and more	0 to 0.3 percent of applied lost per day	0.3 to 4	4 and more	0 to 0.3 percent of applied lost per day	0.3 to 4	4 and more	0 to 0.3 percent of applied lost per day	0.3 to 4	4 and more	0 to 0.3 percent of applied lost per day	0.3 to 4	4 and more
January	straight row	--*	--	--	--	--	--	--	--	--	53	--	--	47	15	--
	terraces	--	--	--	--	--	--	--	--	--	33	--	--	38	1	--
February	straight row	--	--	--	--	--	--	--	--	--	45	--	--	40	11	--
	terraces	--	--	--	--	--	--	--	--	--	22	--	--	27	1	--
March	straight row	--	--	--	--	--	--	--	--	--	37	--	--	35	24	--
	terraces	--	--	--	--	--	--	--	--	--	28	--	--	40	2	--
April	straight row	--	--	--	--	--	--	--	--	--	13	--	--	25	17	--
	terraces	--	--	--	--	--	--	--	--	--	12	--	--	26	1	--
May	straight row	27**	13	--	16	20	--	--	--	--	16	20	3	9	34	13
	terraces	23	11	--	19	15	--	--	--	--	18	18	2	26	23	2
June	straight row	26	--	--	27	1	--	12	7	--	26	2	--	5	20	3
	terraces	3	--	--	10	--	--	11	--	--	14	1	--	11	3	2
July	straight row	4	--	--	8	--	--	18	--	--	20	1	--	2	18	1
	terraces	1	--	--	2	--	--	6	--	--	7	--	--	5	2	1
August	straight row	1	--	--	1	--	--	5	--	--	16	--	--	5	12	--
	terraces	--	--	--	--	--	--	1	--	--	4	--	--	6	1	--
September	straight row	--	--	--	--	--	--	--	--	--	18	--	--	13	6	1
	terraces	--	--	--	--	--	--	--	--	--	5	--	--	4	2	--
October	straight row	--	--	--	--	--	--	--	--	--	10	--	--	6	5	--
	terraces	--	--	--	--	--	--	--	--	--	4	--	--	4	1	--
November	straight row	--	--	--	--	--	--	--	--	--	40	--	--	35	14	--
	terraces	--	--	--	--	--	--	--	--	--	26	--	--	36	3	--
December	straight row	--	--	--	--	--	--	--	--	--	49	--	--	30	19	--
	terraces	--	--	--	--	--	--	--	--	--	34	--	--	32	2	--

* Blank means that there are no pesticide losses during that month

** Number of daily pesticide losses in this category seen in 25 years

TABLE 7-20. MONTHLY FREQUENCY OF PESTICIDE LOSSES IN A 25-YEAR PERIOD UNDER TWO MANAGEMENT PRACTICES IN IOWA

	Group A appl. May 10			Group B appl. May 10			Group B appl. June 10			Group C appl. May 10			Group D appl. May 10		
	0 to 0.3	0.3 to 4	4 and more	0 to 0.3	0.3 to 4	4 and more	0 to 0.3	0.3 to 4	4 and more	0 to 0.3	0.3 to 4	4 and more	0 to 0.3	0.3 to 4	4 and more
	percent of applied per day			percent of applied per day			percent of applied per day			percent of applied per day			percent of applied per day		
January straight row													6		
terraces													14		
February straight row													9		
terraces													9		
March straight row													9		
terraces													12		
April straight row										7			27	13	
terraces										8			25		
May straight row	18*	4	2	16	5	3				11	10	4	7	14	10
terraces	10	3		9	3	1				9	4	1	7	9	1
June straight row	32			37			21	5	1	35	3		9	28	4
terraces	7			11			9	1		15	0		11	4	
July straight row	4			7			10			12			5	6	1
terraces							2			4			4	1	
August straight							8			17			8	8	1
terraces										6			6	2	
September straight row										11			12	3	0
terraces										4			2	1	
October straight row										4			3	2	
terraces										0			2		
November straight row										7			7	5	
terraces										5			10	3	
December straight row										3			11	2	
terraces										3			14	1	

*number of daily pesticide losses in this category seen in 25 years

TABLE 7-21. MONTHLY FREQUENCY OF PESTICIDE LOSS EVENTS FOR 25 YEARS UNDER TWO MANAGEMENT PRACTICES IN NEW YORK

		Group A appl. May 10 percent of applied			Group B appl. May 10 percent of applied			Group B appl. June 10 percent of applied		
		0.0-0.3	0.31-4.0	above 4.0	0.0-0.3	0.31-4.0	above 4.0	0.0-0.3	0.31-4.0	above 4.0
January	straight row									
	terraces									
February	straight row									
	terraces									
March	straight row									
	terraces									
April	straight row									
	terraces									
May	straight row									
	terraces									
June	straight row	1			4			3	1	
	terraces	1			4			3	1	
July	straight row							1		
	terraces							1		
August	straight row									
	terraces									
September	straight row									
	terraces									
October	straight row									
	terraces									
November	straight row									
	terraces									
December	straight row									
	terraces									

TABLE 7-21. MONTHLY FREQUENCY OF PESTICIDE LOSS EVENTS FOR 25 YEARS UNDER TWO MANAGEMENT PRACTICES IN NEW YORK (con't)

		Group C appl. May 10 percent of applied			Group D appl. May 10 percent of applied		
		0.0-0.3	0.31-4.0	above 4.0	0.0-0.3	0.31-4.0	above 4.0
January	straight row				2		
	terraces				2		
February	straight row	1			5	1	
	terraces	1			5		
March	straight row				10		
	terraces				10		
April	straight row				1		
	terraces						
May	straight row						
	terraces						
June	straight row	4	1		1	1	3
	terraces	4	1		1	3	1
July	straight row	1				1	
	terraces	1				1	
August	straight row	3			2	1	
	terraces	2			3		
September	straight row	2				2	
	terraces	2			1	1	
October	straight row	3			2	1	1
	terraces	3			2	1	
November	straight row						
	terraces						
December	straight row	4			5		
	terraces	4			5		

for controlling pesticide loss. For pesticides in Group D, the effect of SWCPs on sediment movement determines pesticide pollution control.

Table 7-22, derived from data in Table 7-17, indicates the percent change in pesticide loss in overland flow in Georgia due to a different pesticide management practice or date of application as compared to the loss of a Group D pesticide applied on May 10 to corn planted either straight row or on terraces. Thus, if a farmer used a Group A pesticide on a terraced corn field, he would obtain a 98% reduction in pesticide loss over a 25 year period as compared to using a Group D pesticide with straight row planting. Table 7-22 also indicates that if a pest could be controlled by a Group B or a Group D pesticide, by applying the former on June 10 as opposed to the latter on May 10, a reduction in loss of pesticide of 99% over a 25 year period for corn planted in straight row would occur.

Other conclusions which can be drawn from Table 7-22 are: 1) substituting a pesticide out of Group A, B, or C for a pesticide out of Group D is more effective in reducing pesticide loss than is implementation of any SWCP; 2) to

TABLE 7-22. PERCENT CHANGE IN PESTICIDE LOSS FOR DIFFERENT PESTICIDE GROUPS, TILLAGE PRACTICES, AND APPLICATION DATES AS COMPARED TO GROUP D PESTICIDES APPLIED ON MAY 10

	Group D (appl. May 10)	
	straight row	terraces
Group A (appl. May 10)		
straight row	-97	-89
terraces	-98	-92
Group B (appl. May 10)		
straight row	-93	-75
terraces	-95	-81
Group B (appl. June 10)		
straight row	-99	-96
terraces	-100	-100
Group C (appl. May 10)		
straight row	-90	-60
terraces	-92	-69
Group D (appl. May 10)		
straight row	---	+74
terraces	-74	---

prevent losses, pesticides should not be applied during a period of high runoff and sediment losses; 3) any combination of the following practices is more effective than is implementing one individually: application of pesticide with a) a shorter half-life, b) a lower adsorption coefficient, c) at a date when probability of runoff is small, or d) implementing management practices which reduce soil and runoff losses. These conclusions are in agreement with findings of a simulation study by Dean and Mulkey (1978).

Similar conclusions can be drawn for Iowa and New York. However, since pesticide losses are not great to begin with in New York (Table 7-15) implementing a SWCP to control pesticide pollution may be of limited value. The conclusions from the simulation studies (Table 7-22) should be considered carefully since they were not verified by field data other than for the base condition.

Pesticide Leaching--

This section describes the sub-model used to simulate the downward movement of pesticide bands over 25 years for two SWCPs at three locations.

The CPM simulates pesticide loss in percolation water as well as in overland flow. As the pesticide band moves downward through the soil, the chemical degrades. Therefore, the model estimates both the depth of the midpoint of the band and the pesticide concentration at this depth.

At some depth in the profile, the pesticide will have degraded to the extent that the original compound can no longer be identified. For the purpose of this section, the lower limit of detection is taken as 0.01% of the amount applied. The average and maximum depth of pesticide bands are shown in Table 7-23.

Several factors affect pesticide leaching: adsorption coefficient -- the stronger the binding, the slower the downward movement, half-life -- the longer the half-life the longer the pesticide remains in the soil, and quantity of water percolating through the soil -- the greater the percolation, the faster the band moves. This effect was seen in Table 7-23. Pesticides of Group A could be detected at a lower depth than Group B because Group A adsorption strength was less. Both Group A and Group B pesticides had the same degradation half-life. Group B and Group C pesticides only differed in half-life (Group C's half-life was approximately seven times as great for Group B). Pesticides of Group C could, therefore, be found deeper in the root zone than Group B.

SWCPs which decrease runoff promote infiltration of water, causing deeper displacement of the pesticide bands. This appears to present a problem, because once a pesticide moves below the root zone, there is little bacteriological activity, and pesticide degradation is greatly hampered. Terraces were found to increase the depth of the pesticide band only slightly. Pesticides did not reach the bottom of the root zone in any simulation (Table 7-23).

In conclusion, there appears to be little danger of pesticide pollution of groundwater in soils where the water table is lower than the root zone.

TABLE 7-23. AVERAGE AND MAXIMUM DEPTH AT WHICH THE AMOUNT OF PESTICIDE IS LESS THAN 0.01% OF THAT APPLIED

	Georgia+		Iowa+		New York+	
	straight row	terraces	straight row	terraces	straight row	terraces
Group A						
average depth (cm)	18	19	14	15	16	16
maximum depth (cm)	28	29	21	23	27	27
Group B						
average depth (cm)	6	6	4	4	6	6
maximum depth (cm)	8	8	6	6	8	8
Group C						
average depth (cm)	26	27	16	16	21	21
maximum depth (cm)	29	31	23	23	28	28
Group D						
average depth (cm)	5	5	4	4	4	4
maximum depth (cm)	5	5	4	4	5	5

+ maximum plant rooting depth for Georgia, Iowa and New York were 75, 135 and 40 cm, respectively

COMPARISON OF PREDICTIONS OF SWCPs

Three models have been used throughout this report. The Agricultural Runoff Management (ARM) Model was used to predict runoff, soil loss, and nutrient and pesticide losses over a ten-year period in Georgia and Michigan. The Cornell Nutrient Simulation Model (CNS) predicted runoff, soil loss, and nutrient losses in overland and percolating waters over a 25-year period in Georgia, Iowa and New York. Finally, the Cornell Pesticide Model (CPM) simulated runoff, soil loss, and pesticide losses in overland and drain water over a 25 year period at the same three locations.

Two management practices were input to all three models: straight row and terraces. Other things in common for all models were runoff and soil loss predictions. Moreover, pesticides in Groups B and D applied on May 10 in the CPM closely resemble the atrazine and paraquat used in the ARM model.

The differences in model predictions can be seen in a summary of results (Table 7-24).

The CPM and CNS models predicted different values for runoff in Georgia. The difference was not expected because both models employed the same runoff formula. One important difference was in the value chosen for antecedent moisture condition (one of the equation variables). This parameter depends on soil moisture content, which the computer continuously budgets. The moisture budgets of the two models differ in zone configuration, evapotranspiration prediction methods, and value of soil moisture content at the beginning of each year.

For Iowa, predicted amounts of runoff are very similar. As opposed to Georgia, antecedent moisture conditions predictions were similar. Percent changes for terraces compared to straight row were all in the same direction. As expected, in New York, the CPM and CNS predictions differed substantially. The CNS model used the SCS curve number for the runoff predictions, while the CPM employed the sub-program which predicted runoff on shallow fragipan soils. The CPM model is valid for soils with good structure, many stones, and hardpans within 50 cm of the surface. The CNS model simulates a field with a deeper hardpan and having a more poorly drained soil. Both of these soil types can be found in New York.

The three models used fairly different methods to predict erosion. The ARM model used the Negev sediment algorithm (Negev, 1967) and assumed chisel tillage. The CNS model used the USLE with the energy term computed by the Onstad-Foster method (Onstad and Foster, 1975). In New York and Georgia, spring plow is assumed, while for Iowa, fall plow is simulated. The CPM model also employs the USLE, but the energy term is computed by the Williams method (Williams and Berndt, 1977). Spring plowing is assumed at all three locations.

Considering all these differences, closeness of the CNS and CPM models' predictions of percent change in soil loss using terraces is somewhat surprising. The ARM model predicted that the SWCP had a smaller effect than was predicted by either the CNS or CPM model.

TABLE 7-24. COMPARISON OF AVERAGE ANNUAL LOSSES FOR ARM, CNS, AND CPM MODELS FOR SELECTED PRACTICES

	Georgia			Iowa		New York	
	CNS[1]	CPM[2]	ARM[3]	CNS	CPM	CNS	CPM
Runoff							
straight row (cm)	22.2	17.1	19	7	6.2	10	2.3
terraces (cm)	20.8	15.8	14.4	5	4.8	4	2.3
% change	-6	-7	-25	-29	-22	-60	0
Soil Loss							
straight row (cm)	24.4	18	3.5	17	6.2	20	0.8
terraces (tons/ha)	9.6	3	2.4	1	0.6	1	0.2
% change	-6	-83	-32	-94	-90	-95	-74
Pesticides							
Group B							
straight row		1.8	1.2				
% of applied							
terraces							
% of applied		1.0	0.9				
% change		-44	-25				
Group D							
straight row		13.6	17.9				
% of applied							
terraces							
% of applied		6.5	10.4				
% of change		-52	-74				

[1] Cornell nutrient simulation

[2] Cornell pesticide model

[3] Agricultural runoff management model

The average annual losses for atrazine and paraquat in overland flow as predicted by the ARM and CPM are again remarkably similar. The percent change in pesticide loss achieved by using the SWCP was not quite the same for these two models, but they were in the same direction.

SUMMARY

The action of SWCPs on unit source areas has been simulated to determine the effectiveness of SWCPs in reducing pesticide loss in runoff and percolating waters. The effects of different pesticide characteristics also were simulated.

Two models were used for the simulation studies, a) the Agricultural Runoff Management (ARM) Model, and b) the Cornell Pesticide Model (CPM) which was developed for this project. Both models had approximately the same precision in predicting pesticide loss.

Simulation runs were made to evaluate the interaction of SWCPs, climate and pesticide characteristics at several locations in the United States for a 10 or 25 year period. The selection of management practices was limited by the capability of the simulation models. Straight row, contouring combined with terraces, sodbased rotations, and minimum tillage were modeled by either one or both models.

The pesticides modeled were grouped as follows:

a) weakly adsorbed and easily degradable
b) intermediately adsorbed and easily degradable
c) intermediately adsorbed and intermediately persistent
d) strongly adsorbed and very persistent

CONCLUSIONS

1. Effects of SWCPs on pesticide loss are not only site specific, but also depend on pesticide characteristics.

2. Greatest pesticide runoff losses occurred in areas with heavy thunderstorms. Therefore, losses in Georgia are much higher than for New York, where they were almost negligible.

3. In all case studies, yearly average pesticide losses were lower for those pesticides with low adsorption coefficients and short half-lives than for these pesticides which were more strongly adsorbed and more persistent.

4. SWCPs are more effective in controlling strongly adsorbed pesticides than in controlling the less well adsorbed pesticides.

5. Under normal agricultural practices, for the pesticides modeled, it is not likely that pesticides will be found in water percolating out of the root zone.

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SECTION 8

COST-EFFECTIVENESS OF SOIL AND WATER CONSERVATION PRACTICES FOR IMPROVEMENT OF WATER QUALITY

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The principal function of soil and water conservation practices has traditionally been to reduce soil erosion on cropland. Recent concern over non-point source pollution from cropland has prompted an interest in examining the effectiveness of SWCPs for improving water quality. In areas where sediment contributes to the degradation of water quality, it has been assumed that practices which reduce soil erosion in a field have the effect of causing less sediment to be available for transport to water. The principal focus in this section is on the cost-effectiveness of SWCPs for control of soil erosion and the transport of sediment to water. Although other substances, primarily pesticides and nutrients, are also potential pollutants, the direct causal relationship between SWCPs and the availability of these substances to water bodies is less clear. Thus investigation of these substances is limited to edge-of-field losses.

Several studies have dealt with the relationship between the cost of implementing SWCPs and their potential benefits to water quality. Studies using a watershed as the basis for analysis have been done by Seitz et al. (1978) in Illinois, Alt et al. (1978) in Iowa, and Taylor et al. (1978) in Texas. Each of these studies compared costs of SWCPs (implementation and maintenance) with assumed benefits. These included both the yield benefits of retaining topsoil and water quality benefits. A number of studies have analyzed costs associated with various farm conservation plans. These studies were conducted in Iowa (McGrann et al. 1978), Pennsylvania (White et al. 1978) and Wisconsin (Sharp et al. 1978). In general, the approach in these studies has been to focus on practices for controlling soil erosion and the reduction in soil erosion was then assumed to result in improved water quality. Only the Wisconsin study evaluated sediment separately from soil erosion and this was done after the fact. That is, farm plans for control of soil erosion were developed, then their effect on sediment reaching waterways was determined. Direct evaluation of the relationship between SWCP implementation and water quality is difficult for a number of reasons including: 1) No generally accepted values exist for the amount of sediment from a field, farm or watershed which should be allowed in a body of water. 2) Existing physical models for predicting sediment transport (Section 5) are not widely accepted and generally involve fairly detailed simulations which make the evaluation of a wide variety of options quite difficult. Thus, at this point, little distinction exists in the literature between methods of implementing SWCPs for control of soil erosion and methods of implementation for the purpose of reducing sediment delivered to streams.

A recent study by Meta Systems (1978) evaluated individual SWCPs at the farm level on the Black Creek Watershed in Indiana. Each practice considered was assumed to be implemented on the whole farm. Estimates of farm cost, soil erosion levels, and sediment delivery to the stream as well as loss estimates for nutrients and pesticides were made. Although useful for comparing individual practices, this study did not compare actual farm plans, where combinations of practices which vary from field to field would typically be used.

A primary objective of this section is to separate and compare the issue of soil erosion from the issue of sediment and its transport to waterways. Soil erosion is undesirable principally because of long-term adverse effects on soil productivity while sediment delivery to waterways is undesirable because of the effects on water quality and sedimentation rates in reservoirs and lakes.

The study evaluates costs and effects of a number of SWCPs and combinations of practices when implemented on individual farms. Costs are interpreted as the loss in net farm income caused by imposing a practice or system of practices on a farm to meet some particular goal such as a reduction in soil loss to some predetermined level. Effectiveness of SWCPs is judged primarily by estimated reductions in sediment reaching a waterway. However, discussions on their effects on soil erosion and on other pollutants are also included. Reducing sediment load depends on both the practice or practices being used, the soil complex, and the particular location of practices relative to the stream. Areas which have a high potential for sediment loss to streams may not have the greatest soil erosion problems. Thus the evaluation of cost-effectiveness of SWCPs for reducing sediment load to streams must consider transport as well as gross soil erosion.

By judging the effectiveness of practices primarily in terms of the reduction of sediment delivered to waterways, the benefits of reduced levels of soil erosion are ignored. It would be expected that when SWCPs are actually implemented on a farm, both types of effectiveness would be considered. Although this section concentrates on the difference between these types of effectiveness, it is not necessarily implied that the goals are mutually exclusive.

Since the emphasis is on examining single practices and systems of practices, in relation to individual situations, the farm was chosen as the unit of analysis. This allows evaluations of the costs of practices as they affect farm operations and thus farm income. Also effectiveness of a practice can be related to the physical properties of the field in which it is implemented.

DEFINITIONS

Most of the terms used in this section have been defined in Section 4. Some additional terms specific to this section are defined below.

Base System. This is the particular cropping system for a farm which is used as the standard for determining both cost and effectiveness of SWCPs on that farm. In general it would designate the most profitable cropping system for the farm without regard to soil erosion or water quality.

Waterway. A waterway is any place where channelized flow would normally occur, including streams, road ditches, sod waterways, diversion ditches, tile outlets, etc. However, for terraces, the waterway begins at the terrace outlet rather than the terrace channel.

Sediment delivery. The average annual amount of sediment reaching a waterway. Units are metric tons per hectare.

Cost of implementing SWCPs. The cost of implementing an SWCP or a system of practices on a farm is the change in expected annual net farm income after implementation as compared to annual income with the base system. Costs generally are expressed in dollars per hectare.

Effectiveness of SWCPs. The effectiveness of an SWCP or combination of practices for control of a pollutant is the expected change in the level of the pollutant reaching a waterway after implementation as compared to the base system. A decrease in the level is considered a positive effect. Units are either kilograms or metric tons per hectare.

Cost-effectiveness of SWCPs. The cost-effectiveness of a practice or system of practices is the effectiveness divided by the cost. Unless otherwise specified, cost effectiveness will always be used in relation to changes in sediment delivery. Units are metric tons per dollar.

Marginal cost of sediment control. At any given level of sediment delivery on a farm, the marginal cost is the additional cost required to reduce sediment delivery by one ton.

OBJECTIVES

1). Summarize factors which affect the cost of SWCPs.

2). Summarize factors which affect the effectiveness of SWCPs for reducing sediment delivery to waterways.

3). Develop some general principles on the cost-effectiveness of SWCPs for reducing sediment delivery.

4). For a few example farms, construct farm plans oriented towards reducing sediment delivery by certain given amounts. Compare these plans with plans which are constructed with an orientation towards reducing soil erosion.

5). Develop representative cost curves which would be facing a farmer attempting to reduce sediment delivery.

6). Analyze how the farm plans for controlling sediment delivery might affect levels of losses of other pollutants.

7). Discuss how current cost-sharing programs might influence the choice of practices used in farm plans for reducing sediment delivery.

METHODOLOGY

Costs

An initial evaluation of the changes in farm costs associated with implementation of SWCPs is done through a general review of the literature. A summarization of this review associates practices with directional changes in individual items in a crop budget.

The magnitudes of these changes are illustrated for an example situation, using a basic crop budget for corn grain from Iowa State University (James McGrann, Personal Communication, 1978). A number of selected SWCPs are budgeted, to illustrate their effects on the items in this basic crop budget. Details on this budgeting process are given in Appendix E.

Effectiveness for Control of Sediment Delivery

To determine the level of effectiveness for selected SWCPs in any given field, both the level of soil erosion and the sediment delivery ratio (SDR) for each practice must be known.

Soil Erosion

Soil erosion was calculated using the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1965). Soil erosion is calculated by the equation $A = R\ K\ L\ S\ C\ P$ where:

A = Average annual gross soil erosion

R = Rainfall erosivity factor for the area

K = Soil erodibility factor for the particular soil

L = Slope length factor

S = Slope gradient factor

C = Cropping practice factor

P = Conservation practice factor

All values for factors used in the equation were from standard sources (either Wischmeier and Smith, 1965 or state SCS offices) except the "C" values for New York and Iowa which were individually calculated, based on recently revised tables (Wischmeier and Smith, 1978).

Soil erosion for terraces included all soil moved down the slope to the terrace channel. Thus terraces affect soil erosion directly only by a change in the L factor, and indirectly as a result of the implied contouring. This is relatively important point since for tile-outlet terraces, only about 10-20% of soil reaching the channel actually leaves the field through the tile (Stewart et al. 1976). Although the sediment deposited in the terrace channel does not actually leave the field, it is eroded from its original location and thus has potential long-term detrimental effects on top soil maintenance. Spreading this material back over the field is difficult and almost never done (Earnest Hintz, Personal Communication, 1978).

For crops such as corn silage which leave little or no crop residue on the field, method of tillage has little effect on soil erosion (Walter Wischmeier, Personal Communication, 1978). Thus the "C" factors for corn silage under conventional tillage, conservation tillage, and no-till are nearly identical.

Estimates of soil erosion by area, soil, and practice and more details on methodology are contained in Appendix E.

Sediment Delivery

In this section, sediment delivery represents the estimated tons of sediment from cropland entering a waterway. Sediment delivery for an area is equal to a calculated sediment delivery ratio (SDR) times the estimated soil erosion as calculated from the Universal Soil Loss Equation. The SDR for an area is a function of the distance from the center of the area to a waterway. For the purpose of this study, a waterway is considered to be an actual stream, a terrace outlet, a diversion ditch, or any other channeled waterway. For tile-outlet terraces, the SDR is multiplied by 0.2 to account for settlement in the terrace channel. Details on the methodology and calculation of SDR are in Appendix E.

It is recognized that in many cases, not all sediment entering a waterway will affect water quality. Some redeposition may occur in the waterway, particularly in grassed waterways and diversion ditches. Also, in some cases, tile outlets and waterways may discharge into woodlands or similar areas which may filter out much sediment. Since these effects are site-specific, they should be taken into account when designing water quality plans for actual farms. The results of this study only apply directly to situations where all sediment entering waterways has a potential effect on water quality.

Of course, other problems are associated with this method for calculation of sediment delivery. Conceptually, the greatest difficulty is that the model implies that the SDR for any area is independent of conditions between that area and the stream. Thus the effect of, say, having a field of sod crop rather than row crop below the field being evaluated is nullified. In particular, grass buffer strips placed along waterways are not evaluated in this analysis. Although claims have been made that buffer stips are effective sediment traps, little verification exists, and their effectiveness is questionable when the width of the strip is considerably less than the total slope length (Section 5).

Despite these shortcomings, the method does seem useful. It should provide a reasonably good way for comparing the relative importance of individual fields or areas of a field relative to their location on the watershed. It also provides a means for comparing relative effectiveness of individual and combinations of SWCPs. Finally, and perhaps more importantly, it may provide a methodology which, when combined with reasonable judgement, could be used in the field for determination of critical areas for control of sediment delivery.

Although the estimates are made in terms of actual tonnage reaching a waterway, it should be emphasized that these may not be good representations of actual amounts reaching the stream. Rather, it is hoped that the relative effects of distance and practice are accurately portrayed. Thus results of the analysis would be better viewed in terms of percent decrease in sediment delivery from the base conditions, rather than as actual amounts of sediment reaching a waterway.

Initial Evaluation of Factors Governing Cost-Effectiveness

To develop some insight into the factors governing the cost-effectiveness of SWCPs, certain individual practices are investigated on a field basis. Soil erosion levels are initially calculated for a hypothetical field with a 4% slope gradient and a 120 m slope. The "R" (rainfall erosivity) factor for the USLE is assumed to be 200, while the "K" (soil erodibility) factor is assumed to be 0.3. These values would fall in a moderately erosive range common for much of the central and eastern United States.

Soil erosion levels are calculated for continuous corn grain with no SWCPs and for each of the practices for which costs budgets were determined in the section on costs.

Initially, the SDR for the field is assumed to be 1.0. This would occur if the field were directly adjacent to a waterway such that all eroded soil entered the waterway. Under these conditions, sediment delivery and effectiveness for individual practices are calculated and compared.

In order to determine the effects of varying SDRs and levels of soil erosion, the effectiveness of the practices is recalculated after first lowering the SDR to 0.3, then by raising the levels of soil erosion by a factor of four. Although these changes are somewhat arbitrary, they serve the purpose of illustrating the relationships between the effectiveness of SWCPs and different types of field situations.

Example Farms and Farm Plans for Control of Sediment Delivery and Soil Erosion

Three areas representing a variety of types of agriculture in the eastern half of the United States were chosen for study. A major soil association is identified for each area. The areas with their representative soil associations are: 1) Eastern Iowa, Tama-Muscatine Soil Association; 2) Central New York, Honeoye-Lima Association; and 3) Eastern Texas, Houston Black-Heiden Association. The example farms are not intended to be "typical" of the area in physical layout of soils, but are idealized versions of each soil association. In each case, the farm is constructed as a rectangle with a stream flowing in the center. The sides of the farm parallel to the stream are considered to be the boundaries of the drainage area such that no water flows onto the farm from outside these areas (See Figure 8-1). Individual soil series (or combinations) are then laid out in strips parallel to the stream with areas approximately proportional to their fraction of the soil associatior Each soil series is assumed to have a uniform slope which is typical of slopes for that series. All land on the farms is assumed to be tillable.

Through contact with state SCS offices and local Extension Service personnel, a list of SWCPs common to each area was determined. These included various tillage practices, rotations, structural practices, and contouring. Because of a lack of data, some practices which may be applicable for the areas studied were not included in the analysis. For example, contour ridge planting appears to be a promising practice for much of Iowa and other cash-grain areas. At present, however, adequate data for budgeting this practice are apparently not available.

Individual crop budgets for each example farm were constructed for both base conditions (no practices) and for each SWCP considered. Fertilizer and pesticide applications rates for each soil and crop budget were based on recommendations made by the Extension Service. Both pesticide and fertilizer (nitrogen) rates are dependent on the crop rotation as well as the individual crop. Available budgets have variable machinery costs and total tractor hours. In certain cases where these figures were not available for all tillage practices, variable machinery costs are calculated from engineering records. Labor requirements are assumed to be 10% greater than tractor hours.

Estimated yields are based on the particular soils and fertilization rates. Effects of SWCPs on yields are variable and depend on the area. For example, in New York it is assumed that neither tillage practice nor crop rotation has an effect on corn yields (Fred Swader, Personal Communication, 1978), while in Iowa a decrease in yields is assumed for no-till while corn following hay or soybeans shows a yield increase (William Shrader, Personal Communication, 1978). In both New York and Iowa, no yield change was assumed for conservation tillage. For these two cases, the budgeting process indicated that conservation tillage was more profitable than conventional tillage. Thus conservation tillage was considered to be the basic tillage system. Conservation and conventional tillage are compared in Table 8-14. Although moldboard plowing is still the predominant primary tillage implement in these areas, the use of other tillage implements appears to be spreading rapidly. Chisel plows and offset discs are now used on nearly 45% of Iowa cropland (James McGrann, Personal Communication, 1978).

Although the farms as laid out for this study would not necessarily have higher costs for contouring, it is assumed that any operations done on the contour require 10% additional machinery and labor time. All fieldwork on terraced fields or fields with diversions is assumed to be done on the contour.

Terraces are included both at SCS recommended spacings and at wider spacings in order to evaluate a range of terrace systems. Although it is not possible to determine all associated costs of wider terrace spacings, it is assumed they would be technically feasible if adequate maintenance of terrace channels is carried out.

Costs for structural practices are amortized costs over a 45-year period. Although this period is somewhat arbitrary, it is within the life expectency for structures in both Iowa and New York. Maintenance costs for terraces and diversions depends on spacing, with wider spacing requiring more maintenaince per unit length of structure.

In each area a cost-share system which paralleled as closely as possible the existing cost-share program was developed. For structural practices where cost-sharing is generally based on a percent of initial construction cost, the cost-share money was subtracted from construction costs before calculating the annual cost. For practices such as strip-cropping, where cost-sharing is often available as a one-time payment based on field area, this payment was amortized over 20 years. Cost-sharing for sod-based cropping systems is generally based on a percent of establishment costs. Thus this could be simply subtracted from the crop budgets.

Detailed explanation of the methods used for determining costs and returns is included in Appendix E.

Development of Farm Plans and Cost Curves

The framework of this part of the analysis is a linear programming (LP) model developed for each example farm. The model is set up to maximize income subject to certain restrictions. These include seasonal labor restrictions, land area, barn capacity, etc. The options include the growing of a variety of crops in combination with the selected SWCPs.

After initially running the model with no restrictions on soil erosion or sediment delivery, the base system for the farm is obtained. The model is then run sequentially with incremental decreases in allowable sediment delivery down to a level of approximately one metric ton per hectare. From these results, farm plans are constructed, representing 50% and 90% reductions in sediment delivery from the base system.

The procedure is repeated using soil erosion as a restraint. Equivalent farm plans are constructed for 50% and 90% reductions in soil erosion. Further analysis and comparison of the results yields information on relative effectiveness of practices and their locations relative to the stream for soil erosion control versus sediment control.

It should be noted that the restraints on sediment delivery were for the average over the whole farm, while soil erosion restraints were on individual fields. Since the major justification for reducing sediment delivery is to improve water quality, the point of origin of the sediment is not important. Thus only whole farm average loss was considered. The justification for controlling soil erosion, however, is to maintain topsoil, thus restrictions on a field basis were considered to be appropriate. This difference in the form of restraints allows more flexibility in developing farm plans for controlling sediment delivery than for controlling soil erosion.

After adding cost-sharing equivalent to current ASCS policies, the model was run again with sediment delivery as the restraint. It is expected that the farmer cost of controlling soil erosion or sediment delivery will be less with cost-sharing. Further analysis should yield methods by which cost-sharing could be made more efficient, particularly in terms of whether the goal of the cost-sharing program is water-quality (control of sediment delivery) or long-term soil productivity (control of soil erosion).

RESULTS

The results are presented in four sections. First, the effects of SWCPs on farm costs and income are investigated. A brief survey of related literature points out the types of effects others have observed. Then a common crop, corn grain, is budgeted for several SWCPs to show the type and magnitude of costs associated with SWCP implementation.

Second, the effectiveness of these practices for controlling sediment loss to waterways is determined for a few hypothetical field situations. Combining this with the cost-budgets already developed, cost-effectiveness values are calculated and discussed.

Three example farms are then evaluated, using an LP model to determine optimal farm plans for controlling sediment loss as compared to soil erosion. Potential effects of cost-sharing programs on these farm plans are also discussed.

An analysis of the relationship between levels of soil erosion, redeposition, and resultant net losses of top-soil for the New York example farm is also presented.

The effects of reduced levels of sediment delivery on nutrient losses is analyzed for the Iowa farm, using results from the CNS model (Section 6).

Finally, the effects of SWCPs on losses of herbicides from the Iowa farm are examined. These results are compared to the reduction in losses that could be achieved by the use of alternative herbicides and by incorporation.

Farm Costs and Benefits of SWCPs

Short-term effects include outlays for structural and maintenance costs, changes in input levels and changes in yields. Items which generally remain unaffected include seed, phosphorus, potassium, lime, harvesting and selling costs, and taxes and insurance. Table 8-1 lists the major cost items likely to change with implementation SWCPs and the likely direction of change for each SWCP. The following review of the literature indicates the reasoning behind the effects noted in Table 8-1.

No-tillage

A number of studies on yield effects of no-tillage corn exist. Using data from two and three year trials, Fink and Wesley (1974) found no significant difference in corn yields between no-tillage and conventional tillage in Illinois. The one exception was in 1970 when heavy rains occurred shortly

TABLE 8-1. POSSIBLE EFFECTS OF SWCPs ON YIELD AND PRODUCTION INPUTS*

	Yield	N	Pesticides	Equipment	Labor	Land	Const. & Maint.
No-tillage	I	0	+	-	-	0	0
Minimum Till	I	0	0,+	-	-	0	0
Rotations	0,+	-	-	0,+	I	0	0
Cover Crops	I	0,-	0,+	+	+	0	0
Contouring	0,+	0	0	0,+	0,+	0,+	0,+
Graded Rows	0	0	0	0,+	0,+	0,+	0,+
Diversions and Grassed Waterways	0,+	0	0	0	0	+	+
Strip Cropping	0,+	0,-	0	0,+	0,+	0	0,+
Terraces	0,+	0	0	0,+	0,+	0,+	+
Ridge Planting	I	0	0,+	+	0,+	0,+	0,+
Listing	I	0	0,+	+	0,+	0,+	0,+
Timing of Field Operations	0,-	0,-	0	0,+	0,+	0	0

*/ + : Increase

- : Decrease

0 : No significant effect

I : Effect may go either way, depending on the situation after planting, causing flooding of the no-tillage slots and the young corn. This reduced the no-tillage plant population; yield was 22% below that for the conventionally tilled plots.

In West Virginia, Bennett et al. (1976) found significantly higher corn yields for no-tillage over conventional tillage following sod. Further evidence that no-tillage following sod helps improve yield relative to conventional tillage was reported by Van Doren and Triplett (1969). In studies conducted in Ohio from 1960-1967 on a clay-loam to clay textured soil, no-tillage yields were 10% below conventional tillage following a row-crop, but less than 4% below on corn following sod. In silt-loam textured soils, corn yields for no-tillage and conventional tillage were approximately equal following a row-crop, but following sod, no-tillage yields were nearly 13% higher than yields with conventional tillage.

A summary of no-tillage yield potentials in Ohio (Forster et al. 1976) indicated yield increases in the neighborhood of 10% on well-drained soils (relative to conventional tillage), approximately equal yields on moderately well-drained and somewhat poorly drained soils, and yield decreases in the order of 10% on poorly and very poorly drained soils.

In general, reports indicate somewhat better results for no-tillage in areas of moderate climate. Underwood (1976) reports 36% higher corn yields for no-tillage over conventional tillage in Virginia while Swader (1970) and Peters (1970) seem to indicate that at best no-tillage yields will equal conventional-till yields in New York and New England. In colder climates, cool spring soil temperatures appear to be a limiting factor in no-tillage corn yields. Tillage buries mulch and aerates soil to facilitate warming. Poorly drained soils warm more slowly than well-drained soils, adding to this problem. Also, since no-tillage corn makes more efficient use of water (Underhill, 1976), well-to excessively drained soils benefit from no-tillage, particularly in dry seasons. Thus spring soil temperature and soil droughtiness are two of the key limiting factors in determining yield effects of no-tillage.

Data on soybean yields under no-tillage are less prevalent. A number of results reported by Clapp (1972) indicate no-tillage soybean yields are comparable to conventionally tilled yields under most situations and may be better in cases where inadequate soil moisture is a problem.

Pest control is another key factor in determining no-tillage yields. Weeds, insects, rodents, birds, and disease can all, in certain situations, be greater problems with no-tillage systems (Phillips et al. 1973). Thus pesticide expenses are generally higher. A contact herbicide, usually paraquat, must be applied when weed size is too great to be controlled by systemic herbicides. There is some evidence that increased use of systemic herbicides, such as atrazine may be necessary for optimal yield levels (Bennett et al. 1976).

Insect problems can also be a greater problem with no-tillage. Armyworms, sod webworms, cutworms, slugs, and root aphids may all cause crop damages (Phillips, 1973). All of these factors add to a generally increased level of pesticide expenditure for no-tillage over conventional tillage.

Although initial conversion to a no-tillage planter requires an increased machinery investment, in the long run this should be offset by selling, not replacing, or possible replacing on a smaller scale tillage equipment such as plows and discs. The situation and attitude of the individual farmer primarily determines which of these courses is followed. Clapp (1972) reports savings of about $9-25/hectare in machinery expenses for soybeans. Fuel savings of up to 73% have been reported for growing corn under no-tillage as compared to conventional tillage. Often the increased costs for pesticides and the decreased machinery and labor costs nearly balance. Estimates from Indiana showed a net savings of only $14-25/ha for no-tillage (Doster et al. 1973). Thus, yield differences appear to be the most crucial factor in determining the profitability of no-tillage practices. Estimated budgets for continuous corn from Ohio State University (Forster et al. 1976) show increased profits of no-tillage over conventional of $29-96/ha on soils where no-tillage yields were greater than conventional - till yields, but decreased profits ranging from $54-103 on soils with lower no-tillage yield potential.

Conservation Tillage

Conservation tillage is generally more suited to colder and wetter soils than no-till (Stewart et al. 1975). Chisel plowing or discing as a primary tillage technique buries roughly 50% of surface plant growth and crop residue, and helps aerate and warm the soil, thus partially overcoming some no-tillage limitations. On moderately well-drained to somewhat poorly drained soils in Ohio, conservation tillage, no-tillage and conventional tillage corn yields were approximately equal. On poorly and very poorly drained soils, conservation tillage yields were slightly better than no-tillage and in some cases even better than with conventional tillage (Forster et al. 1976). In a three-year trial on a moderately well-drained silt loam in Illinois, a chisel-plow tillage system produced average yields about 8% over conventional-till and no-tillage yields. (Fink et al. 1974).

As would be expected, earlier comments on increased expenses for pesticides and savings on machinery and labor inputs for no-tillage operations are somewhat modified with conservation tillage. Although many weeds are buried by chisel and disc tillage, increased weed problems as compared with conventional tillage are still expected (Erbach et al. 1974). Many of the insect problems of no-tillage are also present with minimum-tillage but at a reduced level (Musiek et al. 1973).

Machinery inputs are less than with conventional tillage since chisel or disc plowing is faster and uses less energy than moldboard plowing. Compared to conventional tillage systems, savings of 36% for a chisel-plow system (James McDivett, Personal Communication, 1977) and 46% for a disc tillage system (Wittmuss et al. 1975) have been estimated.

The moderating effects of conservation tillage on net profits as compared to no-tillage are shown in budget estimates by Forster et al. (1976). On two soils, one moderately well-drained and the other somewhat poorly drained, no-tillage profits averaged approximately $38/ha above conventional tillage, while conservation tillage averaged only about $21/ha greater. However, on poorly and very poorly-drained soils, no-tillage average $78/ha less than conventional while conservation tillage averaged $10/ha less.

Sod-based Rotations

The effect of this practice on yields is represented by the difference between yields from row crops grown in sod-based rotations, particularly corn, soybeans, and cotton, as compared to yields from growing these crops continuously. Sod crops may help improve soil structure, organic matter content, and infiltration. The extent to which this is true depends on the soil type and level of management. High clay soils, working soils when too wet, and removing of all crop residue all tend to produce soil conditions which could be improved by a sod crop (S. D. Klausner, Personal Communication, 1977). Under any or all of these conditions, yield increases may be expected for the first one to two years after plowing under sod. Yields of sorghum forage grown on a clay soil in Texas decreased 40% by the fourth year following clover. Soil analysis showed that there was a significantly higher percent of water stable aggregates and organic matter in soils in a 2-year sorghum-sweetclover rotation than in soils growing continuous sorghum (Adams, 1974). Other rotation experiments in Texas clay soils showed cotton yields were considerably higher (19-22%) in cotton rotations with a legume. In rotation with a non-legume, yield ranges were +4% to -17% relative to continuous cotton. Average yields for corn grown in rotations were about 12% above continuous corn (Jetter *et al*. 1962).

On a fine sandy loam in southern Texas, rotations did not have a significant effect on cotton yields. One rotation (cotton, clover, cotton, tomatoes) did, however, apparently somewhat retard development of cotton root rot (Gerard *et al*. 1963). Crop rotations can often be effective in breaking weed, disease, and insect cycles (Stewart *et al*. 1975), particularly for corn rootworm, which can often be completely controlled by using a sod rotation (Luckmann, 1977). Generally, this would imply either a lower pesticide requirement or a higher yield or both.

Savings on nitrogen fertilizer may be considerable in a legume-based rotation. Up to 340 kg/ha nitrogen may be plowed down in a good alfalfa legume crop, of which 170 kg may be available the first year (S. D. Klausner, Personal Communication, 1977). Although other estimates have been somewhat lower, it is generally accepted that legumes can help meet nitrogen requirements for row crops (Stewart *et al*. 1975).

Equipment costs for row crops in a rotation should not change, other than that capital costs for specialized equipment may have to be spread over fewer hectares. However, if a farmer has specialized in continuous row crops, purchase of forage equipment may be necessary to begin crop rotations, adding considerably to machinery costs.

Changes in labor requirements depend completely on the relative labor intensity of the forage crop versus the row crop. Requirements for growing are generally less for hay crops, although harvesting requirements may be higher, particularly for handling dry hay. Even when total labor requirements increased, this may be offset by the more even labor distribution over the growing season.

Cover Crops

Winter cover-crops may have either positive or negative effects on the yield of the succeeding row crop. If excess soil moisture in spring postpones soil warming or field operations, cover crops may help dry fields, thereby allowing timely planting with resultant yield increases. However, under dry conditions, cover crops may draw needed soil moisture, and consequently have an adverse effect on yields (Stewart et al. 1975).

If rye, oats, or other non-legumes are used as a cover-crop, nitrogen requirements for the row-crop are unchanged, but extra N may be needed to facilitate breakdown of cover crop. In areas where a good legume can be established over winter, this may reduce nitrogen fertilizer requirements in the next production period (Mitchell et al. 1977).

In general, pesticide requirements are unchanged from those for conventional tillage. However, with conservation or no-tillage, a contact herbicide may be required to kill the cover crop.

Equipment and labor expenses are increased to cover planting of the cover crop.

Contouring

Since contouring tends to decrease runoff and keep water on the field, yields would be expected to increase where normal soil moisture is inadequate, but be decreased in areas of excess rainfall or poor drainage.

Data reported by U.S.D.A. in 1945 (Stalling, 1945a) from various studies around the country show an average increase in corn yields of about 17% with contouring. Of 31 results reported, only one, from Temple, Texas, shows a yield decrease (28%). The same report shows average soybean yield increases of 14% and average cotton increases of about 12%. No soybean and only one of seven cotton trials showed a yield decrease.

Because of the possibility of short or pointed rows, contouring may require more labor time, particularly in maneuvering multi-row equipment. Land may also be lost in field corners where rows are too short to be useful. On some topographies, contouring is not practical (Stewart et al. 1975).

Graded Rows

All of the cost related to contouring also affect graded rows. Yield effects should be more moderate since excess water is allowed to runoff along the row. Machinery efficiency may be higher than with contouring by grading rows to avoid point rows (Richardson, 1973). Construction of grassed waterways or other outlets may, however, add construction and maintenance costs.

Diversions and Grassed Waterways

When constructed as an SWCP to help control runoff and erosion, diversions and grassed waterways do not necessarily affect yields. However, diversions in particular may help remove excess water allowing more timely field operations and possible yield increases. Some land is lost from production to the diversion and waterway. Construction and maintenance costs are quite varied, depending on the particular situation.

Strip cropping

Since strip cropping always implies rotations and may imply contouring, yield effects of both should occur. Stallings (1945a) shows no consistent effect on yields for several crops. However, many of the reported results were from one year only.

All costs associated with contouring and crop rotations may occur with strip cropping, i.e. nitrogen and pesticide use may decrease, but machinery and labor requirements may increase or stay the same.

Terraces

Like some other practices, terraces can have a positive effect on yield by making more efficient use of water. This would be particularly true in dry areas where level bench terraces are constructed. Little water conservation occurs on gradient terraces unless they are contoured (Stewart, et al. 1975). Since terrace construction disturbs topsoil, initial reductions in yield may occur. However, most terrace construction is on deep soils, often loess. For these soils, yields associated with terraces should eventually equal or surpass non-terraced yields (Spomer et al. 1973).

Stallings (1945a) reports results of the effect of both level and graded terraces on yield. For level terraces, sorghum yields averaged 36% higher, wheat 55% higher, cotton 42% higher, corn 6% higher and oats 4% higher. All are as compared to yields from unterraced fields. The lower benefits for corn and oats are not explained, but are from only one year at Alcester, South Dakota. The other crops were grown in other states and results are for longer time intervals. Stallings notes that all these results are from dry areas (South Dakota, Texas, and Oklahoma) and similar results would not be expected in humid regions.

Results from graded terraces are more varied. Corn yields, as percent of yields from unterraced fields, ranged from 73% in Iowa (3 years) to 126% in Wisconsin (3 years) and 166% in Oklahoma (1 year). Cotton yield increased from 10% to 30%. A 5-year trial in Wisconsin, showed terraced hay yields increased 11% over hay grown on non-terraced soils.

Non-parallel terraces, like contouring, may require additional machinery and labor time. Based on 2 row equipment, Mitchell et al.(1965), estimated that approximately 20% additional field time is required for growing and harvesting soybeans on non-parallel terraces as opposed to flat-land farming. Parallel terraces, however, showed a 10% decrease in field time over non-parallel terraces using four-row equipment for corn production. Construction of level parallel terraces may save machinery and labor time over non-terraced farming if steep slopes or irregular topography create odd shaped fields (Spomer et al. 1973).

Construction and maintenance is the most costly part of terracing, particularly for parallel terraces on uneven topography or terraces requiring grassed back slopes or tile outlets (Stewart et al. 1975).

Ridge Planting and Contour Listing

These practices may increase or decrease yields. Ridge planting is designed for humid areas with excess rainfall, particularly in spring. Ridges warm and dry more quickly and have less problems with excess water during the growing season (Stewart et al. 1975). Contour listing, on the other hand, holds water in the furrows where the crop is planted, thereby improving water use efficiency. Ridging should improve yields under excessive moisture conditions, but may cause decreased yield in dry periods. Contour listing should have just the opposite effect.

Because of possible difficulties in cultivating, either practice may require additional herbicides, depending on the particular situation.

Both practices require some specialized equipment to form the ridges and/or furrows which are generally more expensive than conventional equipment. Like other operations performed on the contour, additional field time may be required for short or pointed rows.

Timing of Field Operations

In areas where fall plowing and fertilizer application are conventional, switching these operations to the spring may cause yield decreases because the planting date may be delayed due to the slower warming and drying of the soil. This is particularly true on fine textured soils with some drainage problems. Research in Iowa and Minnesota shows little or no corn yield gain for fall plowing on well-drained medium textured soils (Swan et al. 1972). However on fine-texture, moderately well to poorly drained soils in southern Minnesota, corn yields on fall plowing averaged 10% greater than yields on spring plowing over a six-year trial. Other results from Illinois (Bateman et al. 1967) and Indiana (Doster et al. 1973) show little difference in corn yields on fall plowing. Since these states are possibly drier than Minnesota and Iowa, early spring drying of the soil may be less advantagous and possibly a disadvantage.

If nitrogen fertilizer is applied in the fall, additional amounts may be needed due to losses by leaching and denitrification (Stewart et al. 1975).

Switching to spring plowing may require additional or larger equipment and require additional spring labor if planting is to be completed by or near the optimal date. Doster (1973) indicates two 70-hp tractors, one 4-16" (40 cm) plow and one 4.3 m disc are adequate for preparing 240 hectares of corn land in central Indiana with fall plowing. On the other hand, spring plowing requires one 140 hp tractor, one 100-hp tractor, one 6-16" (40 cm) plow, one 5-16" (40 cm) plow, and a 7.3 m disc. Total machinery plus labor costs per hectare were estimated at $40.30 for fall-plow and $49.90 for spring-plow.

Some other factors may be actual costs in SWCPs, but are either not well documented, not evident in the short run, or not significant. Most contouring operations require some initial cost in laying out the contours. This generally is done only once, requires minimal equipment, and often is done at no cost to the farm by Soil Conservation Service personnel.

Fertilizer savings, particularly of nitrogen and phosphorous are usually associated with retaining top soil. Presently, fertilizer losses associated with soil and water losses are generally not reflected in most fertilizer recommendations. This is partly because the bulk of these losses are with soil organic matter and are in a form not immediately available to crops. Thus the ultimate effect of these losses is in long-term soil fertility.

Evaluation of Cost-Effectiveness for Selected SWCPs

As indicated in the previous section, costs for implementation of SWCPs generally fall within a range of values and depend primarily upon the crop, the soil, the climate, and the physical properties of the field. Since it is not practical to evaluate all possible conditions, one example situation has been chosen to illustrate the magnitudes of cost-changes associated with some common SWCPs. A budget for continuous, conventionally tilled, straight row corn grain is used as the base condition against which budgets for the SWCPs are compared. This budget is presented in Table 8-2. Note that fixed costs for land, machinery, insurance, etc. are not included. These costs do not change significantly when implementing SWCPs.

Changes in Variable Costs

Table 8-3 shows the changes in variable costs associated with implementing some selected SWCPs. To facilitate comparisons, the columns in 8-3 correspond to the columns in Table 8-1. "Land" is not included since no

TABLE 8-2. SAMPLE CORN GRAIN BUDGET FOR USE IN CALCULATING COSTS OF SWCPs

	Continuous Corn Grain, No SWCPs
Yield MT/ha	7.9
Gross Income, $/ha	695
Variable Costs, $/ha :	
Nitrogen Fertilizer	44
Pesticides	54
Equipment	58
Labor	28
Construction, Maintenance and Other	66
Total Variable Cost	250
Net Income above Variable Costs	$445/ha

TABLE 8-3. TYPICAL CHANGES IN VARIABLE COSTS ASSOCIATED WITH IMPLEMENTATION OF SWCPs AS COMPARED TO CONTINUOUS CORN GRAIN

Values represent expected change in input cost from continuous, straight-row, conventionally tilled corn grain (Table 8-2)

			$/ha			
SWCP	Nitrogen Fert.	Pesticides	Equipment	Labor	Const., Maint. & Other	Total
No-tillage	0	+24	-10	-9	0	+5
Conservation tillage	0	+ 5	- 4	-3	0	-2
Rotation*	-26	-30	-15	-1	-2	-74
Contouring	0	0	+ 6	+3	0	+9
Diversion[#]	-2	- 3	+ 3	+2	+15	+15
Strip Cropping	-26	-30	-11	+2	-2	-67
Terrace A[&]	0	0	+ 6	+3	+101	+110
Terrace B[¢]	0	0	+ 6	+3	+60	+69

* Six year rotation with three years corn, one year oats, two years hay. Values are average for the six years.

[#] One diversion ditch across center of 120 m slope, with contouring. Construction costs are amortized over 45 years.

[&] Terrace A has a terrace at 30 m, 60 m and 90 m, respectively, above lower edge of field with 120 m slope, with contouring. Construction costs amortized over 45 years.

[¢] Terrace B has one terrace at lower edge of 120 m slope, with contouring. Contouring costs amortized over 45 years.

significant amount of land was assumed to be lost for production with any of the practices except the diversion ditch. The diversion required 5% of the land and thus reduced all variable costs by 5% below what they would otherwise be.

For both no-tillage and conservation tillage decreases in machinery and labor expense nearly balance increased pesticide expenditures. Thus the most important factor determining changes in farm income associated with these practices is their effect on yields.

Contouring also shows only a relatively small change in variable costs. However, this practice is not applicable in many fields. Also for this situation, machinery and labor costs for contouring were assumed to increase by 10% above costs for straight-row tillage. Increases of 3 to 4 times this much could occur in many situations.

Changing from continuous corn to a rotation (50% corn) causes a large decrease in variable costs. This is due not only to the lower costs associated with oats and hay, but also to a savings in nitrogen and rootworm insecticide costs amounting to $47.00 for the first year of corn following hay.

Strip cropping costs are simply a combination of rotation costs and costs for contouring. No additional costs would normally occur.

Three systems of structures are considered. To budget these on a per hectare basis, a slope length and gradient had to be specified. These were set at 120 m and 4%, respectively. All field operations with structures are assumed to be done on the contour, thus costs include costs for contouring. Construction costs are amortized over 45 years. The diversion system is one ditch 6 m wide across the center of the field. Since the ditch cannot be used for crop production, all crop inputs are reduced by 5% (as is yield).

The two terrace systems considered are both parallel tile-outlet terraces. Terrace A has a horizontal interval of 30 m which would be approximately equal to SCS specifications. Three terraces are constructed at 30 m , 60 m , and 90 m, respectively, above the lower edge of the field. Terrace B is essentially a sediment basin with one terrace running along the lower edge of the field.

Yield Effects and Total Costs

Some SWCPs can have substantial effects on yields, depending primarily on the climate, soil texture, and soil drainage characteristics. In order to take yield effects into account, total costs have been calculated for these practices for a range of yield levels. These are shown in Table 8-4. Each practice is listed with one or more yield levels. For each yield level, total cost is calculated, where total cost is the change in net income from the base system, conventionally tilled corn grain. A decrease in net income is shown as a positive cost.

TABLE 8-4. EXAMPLES OF TOTAL COSTS ASSOCIATED WITH IMPLEMENTATION OF SWCPs AS COMPARED TO CONTINUOUS CORN GRAIN

SWCP	Assumed* Yield Change %	Corn Grain Yield MT/ha	Gross Income $/ha	Change in Gross Income $/ha	Change in Variable Costs (Table 8-3) $/ha	Total[#] Cost $/ha
No-tillage	+10%	8.7	765	+70	+5	-65
	0	7.9	695	0	+5	+ 5
	-10%	7.1	625	-70	+5	+75
Conservation tillage	+ 5	8.3	720	+35	-2	-37
	0	7.9	695	0	-2	- 2
	- 5%	7.5	660	-35	-2	+33
Rotation[&]	+ 4%[¢]	(**)	524	-171	-81	+90
Contour	+ 5%	8.3	720	+35	+9	-26
	0	7.9	695	0	+9	+9
Diversion[&]	- 5%	7.5	660	-35	+15	+50
Strip Cropping	+ 4%[¢]	(**)	524	-171	-74	+97
Terrace A[&]	0	7.9	695	0	+110	+110
Terrace B[&]	0	7.9	695	0	+60	+60

* Base corn grain yield is 7.9 MT/ha (Table 8-2).

[#] A positive total cost indicates a decrease in net return compared to continuous corn grain with no practices. A negative total cost indicates an increase in net return.

[&] Specifications for these practices are in Table 8-3.

[¢] The 4% yield increase for rotations and strip cropping applies only to corn the first year after hay.

** Rotation and strip cropping yields are: oats, 3.1 MT/ha ; straw, 2.2 MT/ha ; hay, 19.7 MT/ha ; corn, 8.2 MT/ha first year after hay, 7.9 MT/ha second and third years.

The reduced tillage systems show the greatest variance having substantial increases in net income (negative total costs) when yields increase, but also substantial losses in income if yields decrease. Even though the rotation (and strip cropping) showed a large decrease in variable costs (Table 8-3), it still is one of the most costly practices due to the low levels of return on oats and hay as compared to corn.

Except where yield increases are expected, the total cost for many of these practices is a fairly significant proportion of the net income above variable costs with the base crop (Table 8-2). The total cost for what would be considered the standard terrace system, Terrace A, reduces this net income by about 25%. Thus careful consideration of the expected and desired effects should be given before undertaking a program of implementation.

Effectiveness of SWCPs

The effectiveness of SWCPs for reducing sediment delivery to waterways depends on both the initial level of soil erosion and the initial sediment delivery ratio (SDR). To investigate the relative effectiveness of different practices and how effectiveness changes for different field situations, levels of soil erosion and sediment delivery have been calculated for a few hypothetical field situations. Initially, levels of soil erosion were calculated for those practices which were budgeted in the previous section on a field with a 120 m slope and a 4% gradient. These are presented in Table 8-5 in order of decreasing levels of soil erosion. Contouring is assumed for the diversion and the terrace systems. Since the Terrace B system has only one terrace at the lower edge of the field, it acts essentially as a sediment basin and reduces soil erosion little beyond that already accomplished by contouring.

Soil erosion levels for practices such as no-tillage and conservation tillage which depend primarily on crop residue for their effect are sensitive to both crop yield and amount of residue left on the field. The corn yield used here was fairly high, over 30% above the national average. Thus soil erosion levels are low especially for no-tillage. For yields of 5.7 MT/ha , approximately average for the country, soil erosion levels with no-tillage would be about 14 MT/ha , while at yields of 4.4 MT/ha , soil erosion would rise to nearly 21 MT/ha.

To calculate levels of sediment delivery, the SDR must also be known. The SDR is a function of the location of the field relative to waterways and thus is independent of practices on the field except those which cause waterways to be built on or near the field. For this example, diversions and terraces provide waterways in the field and thus may change the SDR, but only for that area of the field lying above the structure. Because of redeposition in the terrace channel, only 20% of sediment reaching the terrace channel is assumed to be lost through the tile-outlet. Thus the SDR for the area above a terrace is 0.2. All sediment reaching a diversion ditch is assumed to have entered a waterway, thus the SDR for areas above diversions is 1.0.

Sediment delivery is initially calculated for two field situations. First an SDR of 1.0 is assumed, which would occur for a field lying directly adjacent to a waterway. Next the SDR is assumed to be 0.3, which corresponds to fields lying roughly in the range of 200 - 300 meters from a waterway. Levels of sediment delivery and effectiveness for the different practices are listed in Table 8-5.

To simplify the discussion on the effectiveness of SWCPs, comparison of the different practices will first be discussed on Field One only (SDR = 1.0). Then the change in effectiveness for each practice when going to Field Two (SDR = 0.3) will be examined.

Since the initial SDR is 1.0, sediment delivery for all practices which do not affect the SDR are the same as the levels of soil erosion. Terraces, however, reduce the SDR and thus are relatively more effective for control-

TABLE 8-5. LEVELS OF SOIL EROSION AND SEDIMENT DELIVERY FOR AN EXAMPLE FIELD[*] AT TWO SEDIMENT DELIVERY RATIOS

		Field One	SDR = 1.0	Field Two	SDR = 0.3
Practice	Erosion	Sediment Delivery	Effectiveness[#]	Sediment Delivery	Effectiveness[#]
		MT/hectare		MT/hectare	
Corn Grain, no SWCPs (Base crop)	31.6	31.6	---	9.5	---
Contour	15.8	15.8	15.8	4.7	4.8
Terrace B[&]	15.5	3.1	28.5	3.1	6.4
Conservation tillage	14.0	14.0	17.6	4.2	5.3
Rotation[&]	13.4	13.4	18.2	4.0	5.5
Diversion[&]	11.2	11.2	20.4	7.3	2.1
No-tillage	8.0	8.0	23.6	2.4	7.1
Strip Cropping[&]	7.9	7.9	24.7	2.4	7.1
Terrace A[&]	7.5	3.0	28.6	1.7	7.8

[*] Slope length = 120 m , Gradient = 4%. USLE factors:
R = 200, K = 0.30.

[#] Effectiveness is the reduction in sediment delivery from the base crop, continuous conventionally tilled corn grain.

[&] Practice specifications are in Table 8-3.

ling sediment delivery than for controlling soil erosion. Also, even though the more intense terrace system (Terrace A) reduces soil erosion to about half the level achieved by the simple terrace system (Terrace B), levels of sediment delivery are about equal for both systems. Since the intense terrace system has three terraces across the slope, but none at the lower edge, the SDR on the lower 25% of the field remains at 1.0, while on the upper 75%, above the terraces, it is only 0.2. Thus the average SDR over the field with this terrace system is 0.4. The other system (Terrace B) has only one terrace, which is located at the lower edge of the field, thus the SDR for the whole field is only 0.2 with this system, or half of what it is with Terrace A. Although the effectiveness of Terrace A could be increased by putting a fourth terrace at the lower edge of the field, this would add significantly to its cost. This option will be discussed further in the section on cost-effectiveness.

The diversion ditch also can change the SDR in a field. In this case, however, the SDR is already 1.0, so no change takes place. Thus, for this situation levels of soil erosion and sediment delivery are the same for the diversion ditch.

In general, for any given field situation, the effectiveness of non-structural practices will be proportional to their respective reductions in soil erosion. For structural practices which decrease the SDR, effectiveness will be more than proportional to reductions in soil erosion, while the effectiveness of practices which increase the SDR will be less than proportional to their decreases in soil erosion.

Table 8-5 shows that the effectiveness of all practices is reduced for Field Two, with an SDR of 0.3, even though levels of soil erosion have not changed. For the non-structural practices, this reduction is directly proportional to the difference in the initial (base) levels of sediment delivery for the two fields. For the terraces and the diversion, the decrease in effectiveness is more than proportional to the difference in the base levels. This can be seen most clearly for the diversion ditch. Since the diversion raises the SDR to 1.0 from its initial level of 0.3 for the area above it, its effectiveness for reducing sediment delivery on this field is below that for all other practices. Sediment delivery is actually higher with the diversion than it would be without it. It is assumed that field operations are done on the contour when a diversion ditch is constructed. Table 8-5 shows that for fields with initially low SDRs, such as Field Two, more sediment is lost with the diversion than with contouring alone.

The terrace systems are also comparatively less effective for controlling sediment delivery than non-structural practices on fields with a low SDR. Terrace B, for example, is more effective than all the non-structural practices on Field One, but less effective than either no-till or strip cropping on Field Two.

Any structural practice which changes the SDR for a field to some fixed number by bringing a waterway to or near the field will be relatively more effective on fields having an initially high SDR. A striking illustration of this can be seen in the following example. A third field, Field Three, with an SDR of 0.3, and with levels of soil erosion four times higher than on Field One is used to illustrate the importance of the SDR in determining the effectiveness of structures. Using Terrace A as an example, levels of soil erosion, sediment delivery, and effectiveness are shown in Field One and Field Three in Table 8-6. Even though the initial levels of both soil erosion and sediment delivery were higher on Field Three than on Field One, a higher degree of effectiveness of the terrace system is obtained by implementing it on Field One because of its initially higher SDR.

TABLE 8-6. EFFECTIVENESS OF A TERRACE SYSTEM IN REDUCING SEDIMENT DELIVERY ON TWO EXAMPLE FIELDS

	Field One SDR = 1.0 MT/ha			Field Three SDR = 0.3 MT/ha		
	Soil Erosion	Sediment Delivery	Effectiveness*	Soil Erosion	Sediment Delivery	Effectiveness
Corn Grain (Base Crop)	31.6	31.6	---	126.4	37.9	---
Terrace A[#]	7.5	3.0	28.6	30.0	6.8	23.2

* Effectiveness is the change in sediment delivery with implementation of SWCP

[#] Terrace A is specified in Table 8-3.

Although the above example was hypothetical and the relationship would not necessarily be the same for other structures or even other terrace systems, it does illustrate the point that simply looking at levels of soil erosion does not necessarily indicate those fields or areas for which SWCPs would be most effective in reducing sediment delivery.

The final point on practice effectiveness deals with the optimal placement of crops on different areas of a farm or field depending upon the SDR for each area. In general, maximum effectiveness in reducing sediment delivery is obtained if crops or SWCPs with the lowest associated levels of soil erosion are placed in areas with the highest SDRs. This is illustrated in Table 8-7. Using Field Two with an initial SDR of 0.3 as an example, it was assumed that contouring was implemented with a diversion ditch constructed across the center of the field. This reduces average levels of soil erosion by about 73%. The upper half of the field, above the diversion, now has an SDR of 1.0, while the SDR on the lower half remains at 0.3. If half the field were to be put into strip cropping which has a relatively low level of soil erosion, the other half into contoured corn grain which has a considerably higher level of soil erosion, significant differences in reductions in sediment delivery are obtained depending on where the two crops are placed, even though placement has no effect on the level of soil erosion. Table 8-7 shows that placing the strip cropping above the diversion and contoured corn below (Option Two) increases the reduction in sediment delivery (effectiveness) by nearly 70% over reversing this placement, even though no change in the level of soil erosion occurs.

TABLE 8-7. OPTIMAL PLACEMENT OF CROPS FOR REDUCING SEDIMENT DELIVERY IN A FIELD WITH A DIVERSION DITCH

	Field Two (initial SDR = 0.3)		
	Soil Erosion	Sediment Delivery	Effectiveness
	MT/ha		
Corn Grain (Base Crop)	31.6	9.5	---
Option One: Contoured Corn above diversion, strip cropping below diversion	8.4	6.4	3.1
Option Two: Strip cropping above diversion, contoured corn below diversion	8.4	4.5	5.0

Cost-Effectiveness of SWCPs

When a goal has been set to reduce the levels of sediment reaching waterways, and there is a cost associated with reaching the goal, then it is desirable to have some measure of the efficiency of plans or practices which may be implemented to achieve the goal. Cost-effectiveness, defined as the reduction in sediment delivery achieved per dollar of cost, is one measure of the efficiency of practices, combinations of practices, or total farm plans for reaching the goal of reduced sediment delivery. In the previous two sections, costs and levels of effectiveness were determined for a number of SWCPs under certain specified conditions. In Table 8-8, these costs and levels of effectiveness are combined to determine cost-effectiveness values for SWCPs. The cost-effectiveness values were calculated for the two field situations used to investigate effectiveness (Table 8-5), and with the various levels of change in yields used in calculating costs (Table 8-4). In situations where yield increases would occur when implementing either no-tillage, or contouring or where they would not decrease when changing to conservation tillage, increases in net income (negative costs) occur. Cost-effectiveness is not defined for these situations since it is assumed they would be adopted for economic reasons and thus would be part of the base system for a farm.

TABLE 8-8. COST-EFFECTIVENESS VALUES FOR SELECTED SWCPs FOR TWO EXAMPLE FIELDS

Practice	Assumed % Yield Change	Cost[#] ($/ha)	Field One, SDR = 1.0[*] Effectiveness[&] (MT/ha)	Field One, SDR = 1.0[*] Cost-Effectiveness (MT/$)	Field Two, SDR = 0.3[*] Effectiveness[&] (MT/ha)	Field Two, SDR = 0.3[*] Cost-Effectiveness (MT/$)
Contour	+5%	-26	15.8	(**)	4.8	(**)
	0	9	15.8	1.76	4.8	.53
Terrace B[¢]	0	60	28.5	.48	6.4	.11
Conservation tillage	+5%	-37	17.6	(**)	5.3	(**)
	0	-2	17.6	(**)	5.3	(**)
	-5%	+33	17.6	.53	5.3	.16
Rotation[¢]	+4%	+90	18.2	.20	5.5	.06
Diversion[¢]	-5%	+50	20.4	.41	2.1	.04
Strip Cropping[¢]	+4%	+97	24.7	.25	7.1	.07
Terrace A[¢]	0	110	28.6	.26	7.8	.07
No-till	+10	-65	23.6	(**)	7.1	(**)
	0	+5	23.6	4.72	7.1	1.42
	-10%	+75	23.6	.31	7.1	.09

[*] Field One and Field Two have the same levels of soil erosion and differ only in their SDR. Field specifications are in Table 8-5.

[#] Cost is defined as the loss in income with the SWCP as compared to continuous corn grain with no SWCPs. See Table 8-4.

[&] Effectiveness is defined as the drop in sediment delivery with the SWCP as compared to continuous corn grain with no SWCPs. Initial levels are 31.6 MT/ha on Field One and 9.5 MT/ha on Field Two. See Table 8-5.

[¢] Specifications for practices are in Table 8-4.

[**] Cost-effectiveness is not defined for practices which show an increase in net return (negative cost).

Many of the factors affecting the cost-effectiveness of practices were discussed in the section on costs and the section on effectiveness. Thus some repetition is involved in analyzing cost-effectiveness. Also, as previously mentioned, there is a fair amount of variance in both the cost and the effectiveness of these practices in other field situations. Thus the values of cost-effectiveness and their rank do not necessarily remain constant for all cases. However, a number of conclusions can be reached, using Table 8-8 as a guide.

It would appear that the practices with the highest potential levels of cost-effectiveness are no-tillage, contouring, and conservation tillage. However, there are many fields where these high levels cannot be achieved. Yield losses will often occur with no-tillage in areas with poorly drained or finely textured soils or which typically have cold spring soil temperatures. Yield losses greatly reduce the cost-effectiveness of no-tillage. Contouring is only practical in areas with a reasonably even topography. Although Table 8-8 does not show a particularly high value for the cost-effectiveness of conservation tillage, it may be adaptable in more situations than either no-tillage or contouring. Under most conditions, yield levels can be maintained at least equal to conventional tillage. Thus, often little or no cost is associated with this practice.

Switching to a sod-based rotation is one of the least cost-effective practices shown in Table 8-8. However, this value also can vary considerably. Hay markets are generally local rather than national, thus prices vary considerably. The price used here was $37/MT, which is based on average current prices in Iowa. In some areas, particularly the Northeast, prices have been nearly double that amount in recent years. Also, most dairy and beef farms grow some hay to obtain a balanced feed ration. The cost of increasing the amount of hay grown for this type of farm may be less than for a cash-crop farm, since the hay can be utilized on the farm rather than being sold.

The cost-effectiveness for strip cropping shown in Table 8-8 is also quite low. However, it is based on changing from continuous corn grain to strip cropping. In situations where a sod-crop is already being grown and thus strip cropping can be implemented with no increase in the area of sod crops, it will often be one of the most highly cost-effective practices available. In this situation, the cost would be approximately equal to that for contouring, and the effectiveness (and thus the cost-effectiveness) about 50% greater than for contouring.

The two terrace systems have nearly equal levels of effectiveness, however, Terrace B having only one terrace along the foot of the slope is considerably more cost-effective than Terrace A, which has terraces at 30-m intervals starting 30 m above the foot of the slope. It was noted in the previous section that the effectiveness of Terrace A system could be increased by adding one more terrace at the foot of the slope. This situation is evaluated in Table 8-9. There is an increase in costs, a slight increase in effectiveness and a slight decrease in cost-effectiveness for this system as compared to Terrace A. However, a fourth terrace system, Terrace D, evaluated in Table 8-9 has three terraces, like Terrace A, but space at 40-m intervals with one terrace at the lower edge of the field. Terrace A and Terrace D have the same cost, Terrace A has a lower level of soil erosion, but Terrace D is most effective for reducing sediment delivery and thus has the higher cost-effectiveness. Terrace B is still the most cost-effective structural practice. However, there may be a question on the technical feasibility of placing just one terrace on such a long slope. It is possible that supporting practices such as no-tillage or strip cropping would be required in some field situaitons. More research in this area may be required.

TABLE 8-9. COST, EFFECTIVENESS, AND COST-EFFECTIVENESS OF A MORE INTENSE TERRACE SYSTEM

			Field One SDR = 1.0		Field Two SDR = 0.3	
	Cost ($/ha)	Soil Erosion (MT/ha)	Effectiveness (MT/ha)	Cost-Effectiveness (MT/$)	Effectiveness (MT/ha)	Cost-Effectiveness (MT/$)
Terrace C*	145	7.4	30.1	.21	8.0	.06
Terrace D#	110	8.7	30.0	.27	7.8	.08

* Four terraces at 30-m intervals starting at the foot of the slope.

# Three terraces at 40-m intervals starting at the foot of the slope.

All the practices have lower values of cost-effectiveness on Field Two than Field One. Except for the terraces and diversions, the levels of cost-effectiveness for any individual practice on the two fields are proportional to the initial levels of sediment delivery for the two fields. This relationship is independent of the individual SDR or the level of soil erosion for either field. However, comparisons of this type are not valid if the cost of implementing a practice varies considerably between two fields.

This type of relationship also holds for structural practices. However, as discussed in the section on effectiveness, the cost-effectiveness of structures depends on both the initial SDR and the initial level of sediment delivery. Thus if two fields have identical levels of sediment delivery, but different SDRs and levels of soil erosion, structures will be more cost-effective on the field with the higher SDR (and thus the lower level of soil erosion). Non-structural practices would have the same cost-effectiveness on both fields. Table 8-8 shows a ratio of about 3.3:1 for levels of cost-effectiveness of each non-structural practice on Field One versus Field Two. However, Terrace A, Terrace B, and the diversion have ratios of 3.7:1, 4.4:1, and 10.2:1, respectively, reflecting their relatively lower levels of cost-effectiveness on Field Two which has the lower SDR.

The cost-effectiveness of a combination of practices will generally be less than for either practice by itself. Costs for combinations are usually simply the sum of individual costs. Effectiveness, is usually proportional to existing levels of sediment delivery. Thus, for instance, if two practices both reduce sediment delivery by, say, 50%, and both have the same cost, then in combination, their cost is the sum of the individual costs or twice the cost of either by itself. The effectiveness however, is only 1.5 times the individual effectiveness of either practice. Thus the cost-effectiveness of the combination is just 75% (1.5/2) of the cost-effectiveness of either individual practice. Thus combining practices tends to reduce cost-effectiveness. In cases where two practices are combined which individually have significantly different values of cost-effectiveness, the cost-effectiveness of the combination may be higher than for the practice which originally had the lowest value, but will always be less than the simple average of the two individual cost-effectiveness values. For example, the cost-effectiveness of combining no-tillage (with no assumed yield change) with contouring on Field One (Table 8-8) is 2.06 MT/$, higher than for contouring alone, but lower than the average of the two practices (3.49 MT/$). The result of this relationship is that when high degrees of effectiveness are desired such that practice combinations are required, costs will increase proportionately more than the decreases in sediment delivery. Another way of viewing this conclusion is to note that if a certain amount of money were available to invest in reducing sediment delivery, it would be more effective to attempt to achieve moderate reductions over a wide area than to concentrate on large reductions in a smaller area.

Farm Plans for Controlling Sediment Delivery and Soil Erosion on Three Example Farms

The three example farms represent different types of farming operations in three parts of the country: New York, Iowa, and Texas. The farms are laid out as idealized models of a typical soil association in the areas being studied. Each is a rectangle with a stream flowing through the center. Individual soils of the soil association are in strips parallel to the stream, positioned as they would typically occur on a watershed. A list of the soils for each farm, their slopes, slope lengths, and distances from the stream are given in Table 8-10. Each soil type was considered an individual field on which one or more crops could be grown.

The discussion on each example farm concentrates on looking at a farm plan for reducing sediment delivery by 50%, and then a plan for obtaining a 90% reduction. Because of the importance of practice and crop location in farm plans for reducing sediment delivery these plans are illustrated as well as described. The plans are compared to plans for achieving similar reductions in soil erosion. In general, the cost of reducing soil erosion by a given percentage is greater than that for achieving the same percentage reduction in sediment delivery. This is due to a number of factors, including:

1). Restrictions on soil erosion are placed on each individual soil type, while restrictions on sediment delivery apply only to the average over the whole farm.

2). Changing the location of individual crops or practices will often have a greater effect on sediment delivery than on soil erosion.

3). Terraces are more efficient at reducing sediment delivery than soil erosion, particularly when placed in areas with high sediment delivery ratios.

Thus in general, there is more flexibility in designing a farm plan for reducing sediment delivery than in designing a plan for reducing soil erosion.

New York Example Farm

The New York example farm is a dairy farm with a maximum of 80 cows and 70 crop hectares. Three crops are grown, corn silage and hay to feed the dairy herd, and corn grain for either feed or sale.

TABLE 8-10. DESCRIPTION OF SAMPLE FARMS

Soil Type	Size (Ha)	Slope Gradient (%)	Slope* Length (Meters)	Distance# to Stream (Meters)
	New York: Honeoye - Lima Association			
Honeoye-Lima	35	8.0	250	250
Lima-Kendaia	35	4.0	250	0
Totals	70	---	500	---
	Iowa: Tama - Muscatine Association			
Muscatine	30	1.0	150	350
Tama	40	3.5	200	150
Dinsdale	15	7.0	75	75
Garwin	15	1.0	75	0
Totals	100	---	500	---
	Texas: Houston Black - Heiden Association			
Houston Black	45	2.0	110	140
Heiden	35	3.0	90	50
Trinity	20	0.5	50	0
Totals	100	---	250	---

* Slope length is given for each individual soil, however, soil loss calculations were made using the total slope length and apportioning soil loss to individual fields by the method of Foster and Wischmeier (1973).

# Distance from lower edge of soil type to the stream.

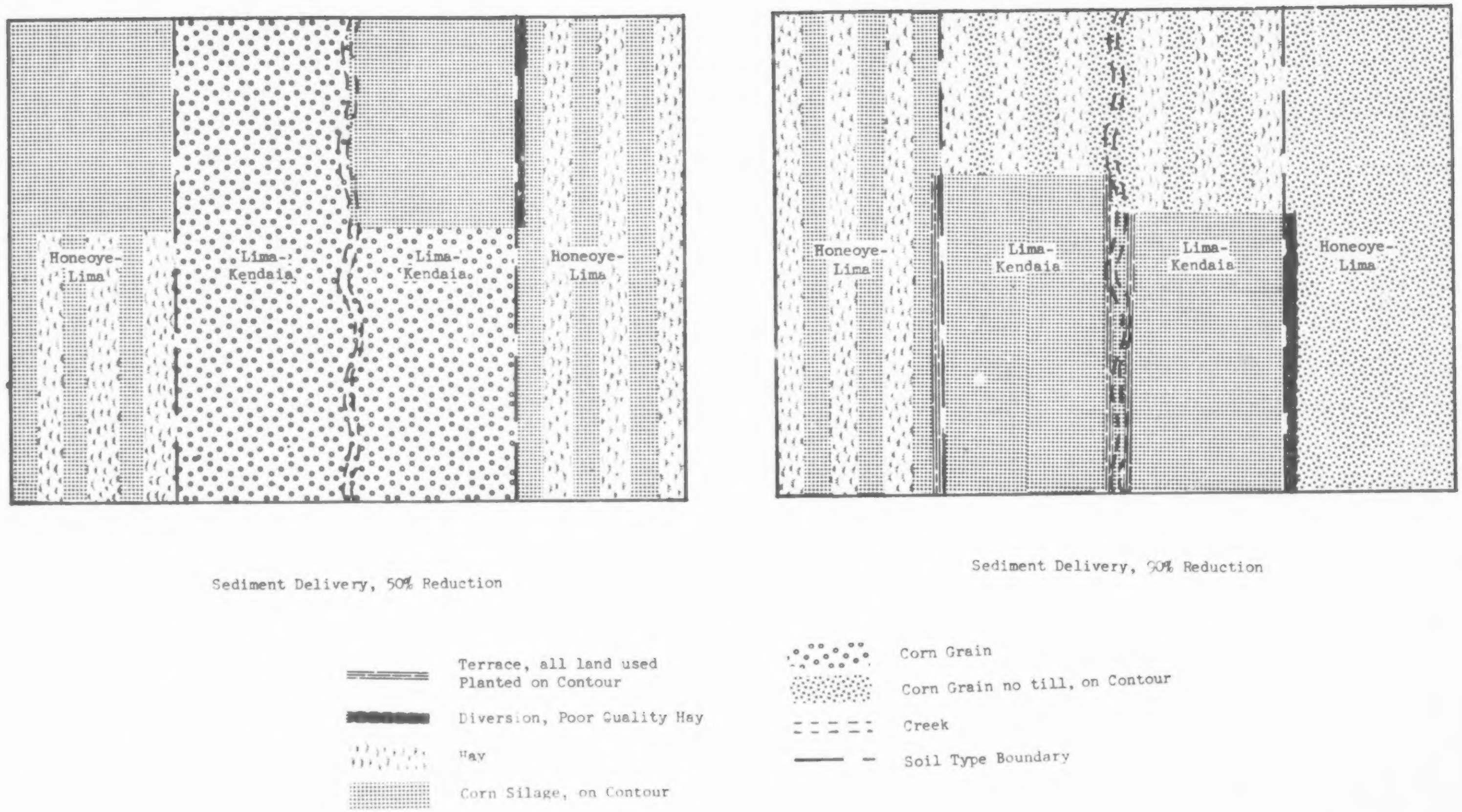
The base system (no restrictions on soil erosion or sediment delivery) consists of 35 ha of corn silage on the Lima-Kendaia soil which lies next to the stream, and 20 ha of corn grain in rotation with 15 ha of hay on the Honeoye-Lima soils. The base plan also assumes the use of a chisel plow (conservation tillage). No expected yield difference occurs between conventional and conservation tillage on these soils, thus conservation tillage is the most profitable tillage method (see Table 8-14). Farm plans for obtaining a 50% and a 90% reduction in sediment delivery are shown in Figure 8-1. Levels of sediment delivery and soil erosion for the base system and each farm plan are given in Table 8-11. Note that due to redeposition, the levels of soil erosion shown in Table 8-11 do not necessarily indicate the net loss of soil from either field. Net losses of soil for this example farm are discussed in a later section. Under the base system, levels of soil erosion and sediment delivery are extremely high on the lower field (Lima-Kendaia soil). This field lies on the lower half of a rather long slope and receives considerable runoff from above. Combined with a cropping pattern of continuous corn silage, high levels of soil erosion result. Since this field also lies next to the stream, it has a high SDR. Thus the Lima-Kendaia soil is the most critical area on this farm for controlling sediment delivery and soil erosion.

To achieve a 50% reduction in sediment delivery (Figure 8-1) all corn grain production is switched to the lower field (Lima-Kendaia soil). Corn grain has considerably lower levels of soil erosion than corn silage. A small area on this soil remains in corn silage, however, it is contoured and lies below a diversion ditch. The diversion ditch reduces the effective slope length and thus reduces levels of soil erosion on this area. Since the diversion ditch increases the SDR on the upper field (Honeoye-Lima), the area immediately above the ditch is strip-cropped to minimize soil erosion. Note that if the contoured corn silage in the upper field were above the diversion ditch, instead of strip cropping, sediment delivery would be increased, but neither soil erosion nor farm income would be changed.

A 90% reduction in sediment delivery is achieved by adding a simple terrace system and switching to no-tillage for corn grain production. Since almost all crop residue is removed with corn silage, no-tillage has little effect on reducing soil erosion and thus is not used on this crop. Crop location is again a key to obtaining this level of reduction. Continuous contoured corn silage, the most erosive crop in this plan, is grown above a terrace, which acts to trap sediment, and below either a terrace or a diversion ditch, which reduce the slope length. Most of the streambank is protected by a terrace. Where terraces along the streambank are not built, strip cropping with no-tillage corn grain is implemented. This is the least erosive cropping system available and thus minimizes soil erosion in this area which has a high SDR.

On the upper field, contoured no-tillage corn grain is placed above the diversion. This is a less erosive system than strip cropping with silage, which is thus grown above the terrace.

The key to the location of all these practices is an attempt to place the most highly erosive crops in areas with the lowest SDR, and the least


Honeoye-Lima
Lima-Kendaia
Lima-Kendaia
Honeoye-Lima
Honeoye-Lima
Lima-Kendaia
Lima-Kendaia
Honeoye-Lima
Terrace, all land used Planted on Contour
Diversion, Poor Quality Hay
Hay
Corn Silage, on Contour
Corn Grain
Corn Grain no till, on Contour
Creek
Soil Type Boundary

Sediment Delivery, 50% Reduction

Sediment Delivery, 90% Reduction

FIGURE 8-1. FARM PLANS FOR REDUCING SEDIMENT DELIVERY, NEW YORK.

erosive crops in areas with higher SDRs. Since this generally has little effect on soil erosion, it is one of the major differences between plans for reducing sediment delivery and plans for reducing soil erosion. A second option used in the sediment delivery plans is the placing of a terrace along the streambank. This also has a minimal effect on soil erosion. Thus, farm plans for reducing soil erosion must rely on other practices which are often more expensive. This is particularly true at high levels of reduction. Table 8-11 shows that obtaining a 90% reduction in soil erosion costs about four times as much as obtaining an equivalent percentage reduction in sediment delivery. The major expense in the farm plan for a 90% soil erosion reduction is a system of eight diversion ditches running the full length of the farm (parallel to the stream) at 60-m intervals.

There is not a great difference in the farm plans for reducing soil erosion and sediment delivery at a reduction level of 50%. The diversion ditch is about 50% longer for the soil erosion plan and most grain is grown by the no-tillage method. These differences account for the slightly greater cost associated with the soil erosion plan (Table 8-11).

Iowa Example Farm

The Iowa example farm is a 100-hectare hog-cash crop farm. Corn grain could be grown either to feed hogs or to sell while soybeans, oats, and hay were grown only as cash crops. Soybeans could be grown only in rotation with corn. As in the New York case, conservation tillage was the most profitable tillage system and thus was used in the base system (see Table 8-14). Two options available in New York were not used in Iowa. Yield losses of around 5% are expected with no-tillage, thus this practice is not common

TABLE 8-11. NEW YORK: FARM PLANS FOR REDUCING SEDIMENT DELIVERY AND SOIL EROSION

		Plans for Sediment Delivery		Plans for Soil Erosion	
	Base System	50% Reduction	90% Reduction	50% Reduction	90% Reduction
Sediment Delivery (MT/ha)					
Honeoye-Lima	5.1	8.5	2.7	7.1	3.4
Lima-Kendaia	47.1	17.5	2.3	14.9	4.0
Farm Total	26.1	13.0	2.5	11.0	3.7
Soil Erosion (MT/ha)					
Honeoye-Lima	16.9	24.8	7.4	19.0	4.0
Lima-Kendaia	60.2	22.3	11.7	19.0	4.0
Farm Total	38.5	23.6	9.6	19.0	4.0
Cost ($/ha)	---	$5.13	$13.01	$7.21	$52.77

and was not included. Diversion ditches are also seldom used in this area of Iowa and were not considered. Both fall and spring tillage were considered. It is fairly difficult to accurately determine the economic costs of spring tillage, although they certainly exist. To account for these costs, an arbitrary additional labor charge of $0.50/ha was assigned to all spring tillage work. The farm plans for Iowa shown in Figure 8-2 do not distinguish between fall and spring tillage. However, as restrictions on sediment delivery become greater, more tillage is shifted to the spring. For the base plan, over 90% of tillage is in the fall, at a 50% reduction, nearly one half of tillage work is in the spring, and at 90% reduction level, about 60% of tillage occurs in the spring.

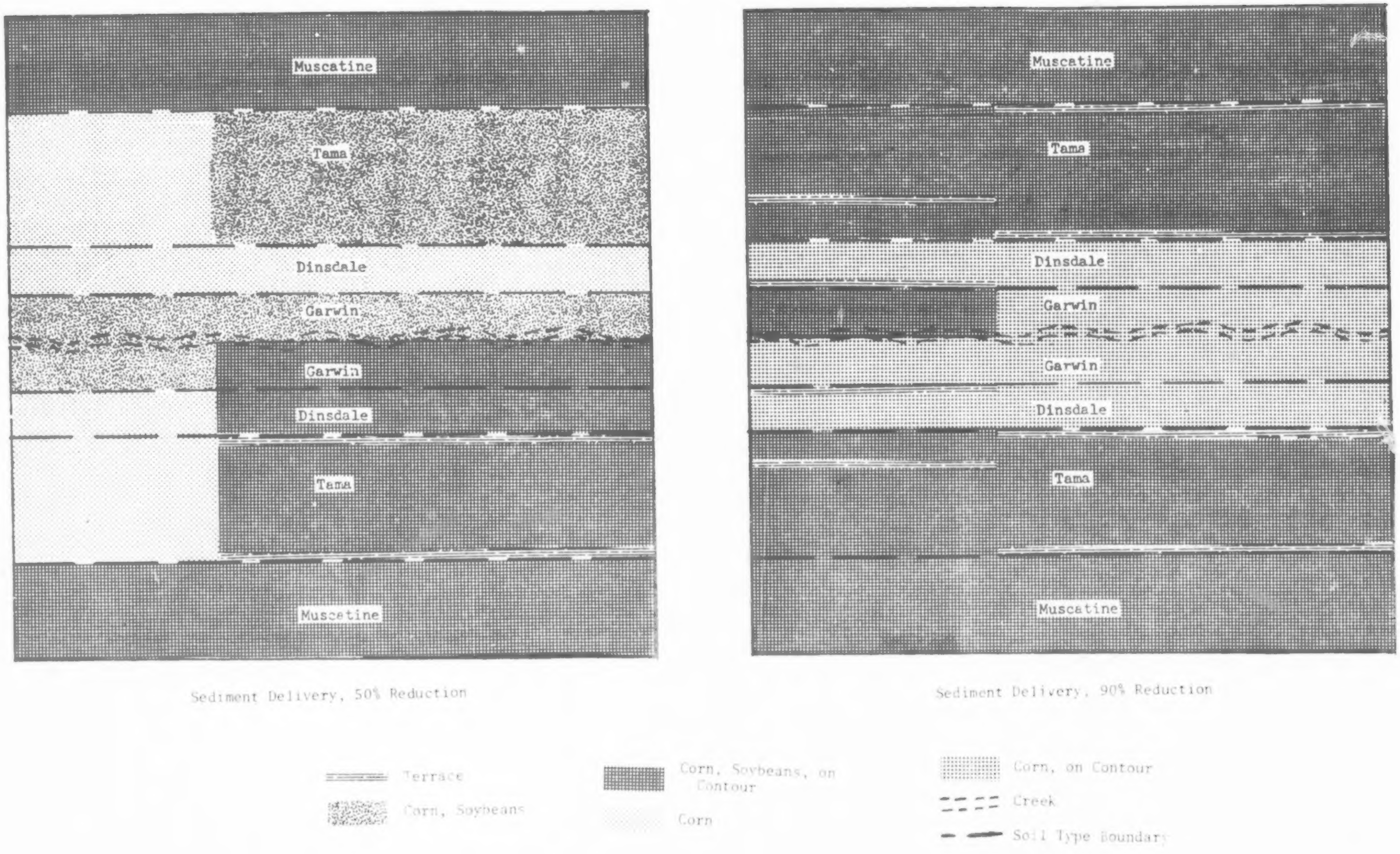
The base system in Iowa consisted of 20 ha of continuous corn grain and 80 ha of the corn-soybeans rotation. The continuous corn is grown on the Tama soil, while the more erosive cropping pattern, corn-soybeans is grown on the other soils, including the Dinsdale and Garwin which lie closest to the stream. Levels of sediment delivery and soil erosion for the base system and the farm plans are shown in Table 8-12.

The farm plan for reducing sediment delivery by 50% utilizes a two-terrace system covering about one third of the farm. One terrace runs along the lower-edge of the Muscatine soil, the other along the lower edge of the Tama soil. A single terrace placed on a field reduces the SDR for the area above the terrace to 0.2 but does not affect the level of soil erosion on that area (except for the contouring effect). Conversely, the terrace reduces soil erosion on the area below the terrace by reducing the runoff from above, but does not affect the SDR on this area. Thus levels of soil erosion on the areas of Dinsdale and Garwin below the terrace system are reduced. Since the Dinsdale soil has the highest levels of sediment delivery in the base plan, it is the most critical area to control. The area of Dinsdale below the terrace remains in a corn-soybeans rotation but is contoured and spring-plowed. The area of Dinsdale not below the terrace is switched from fall-plowed corn-soybeans to spring-plowed continuous corn, reducing soil erosion levels in this area. The total area of continuous corn is increased to about 24 ha with this plan.

The plan for a 90% reduction in sediment delivery has the same terrace system on about 50% of the farm. On the remaining 50% a two-terrace system is implemented with one terrace across the Tama soil and one terrace at the foot of the Dinsdale soil. This latter system is more efficient at reducing sediment delivery than the former terrace system, but is more expensive to maintain due to the longer slope lengths involved.

All of the Muscatine and Tama soils are in a corn-soybeans rotation. The Dinsdale soil is completely in continuous corn while the area of Garwin below the first terrace system is in continuous corn. These areas have the highest potential for soil erosion and sediment delivery. The total area of continuous corn remains at 24 hectares for this plan, with the remainder in the corn-soybeans rotation.

The farm plan for a 50% soil erosion reduction uses the same terrace system as the plan for reducing sediment delivery by 50%. However, in order


Muscatine
Tama
Dinsdale
Garwin
Garwin
Dinsdale
Tama
Muscatine
Muscatine
Tama
Dinsdale
Garwin
Garwin
Dinsdale
Tama
Muscatine
Terrace
Corn, Soybeans
Corn, Soybeans, on Contour
Corn
Corn, on Contour
Creek
Soil Type Boundary

Sediment Delivery, 50% Reduction

Sediment Delivery, 90% Reduction

FIGURE 8-2. FARM PLANS FOR REDUCING SEDIMENT DELIVERY, IOWA.

to reduce soil erosion on the Dinsdale soil, this area is completely in a corn-oats-hay rotation. This results in the increased cost over the sediment plan.

To obtain a 90% reduction in soil erosion, over 70% of the farm has a system of four terraces and nearly 25% is in a corn-oats-meadow rotation. Thus costs are greatly increased over the plan for reducing sediment delivery by 90% (Table 8-12).

If farm plans are designed to reduce soil erosion, crop placement may be highly unfavorable for controlling sediment delivery. The Iowa plan for a 90% reduction in soil erosion is a good example of this. All of the Garwin soil, which lies next to the stream and has an SDR of 1.0, is in the highly erosive corn-soybeans rotation, while most of the hay rotation is in areas directly above terraces, which have a low SDR.

Although Table 8-12 clearly shows that for the Iowa example farm, it is less costly to achieve reductions in sediment delivery than to achieve percentage equivalent reductions in soil erosion, it also shows that each soil erosion plan reduces sediment delivery proportionally more than it reduces soil erosion. When soil erosion is reduced by 50%, sediment delivery actually decreases by 74% (to 5.0 MT/ha). The cost is $29.56/ha. However,

TABLE 8-12. IOWA: FARM PLANS FOR REDUCING SEDIMENT DELIVERY AND SOIL EROSION

	Base Plan	Plans for Sediment Delivery 50% Reduction	Plans for Sediment Delivery 90% Reduction	Plans for Soil Erosion 50% Reduction	Plans for Soil Erosion 90% Reduction
Sediment Delivery (MT/ha)					
Muscatine	1.5	0.8	0.6	0.9	0.6
Tama	8.5	5.8	1.7	5.6	0.6
Dinsdale	68.3	27.5	2.9	8.1	1.4
Garwin	9.1	6.9	2.1	7.5	1.4
TOTAL FARM	15.5	7.8	1.5	5.0	1.1
Soil Erosion (MT/ha)					
Muscatine	5.0	3.0	3.0	4.4*	3.0
Tama	20.8	15.9	8.4	15.0	3.0
Dinsdale	128.9	51.1	10.3	15.0	3.0
Garwin	9.1	6.9	2.9	7.5	3.0
TOTAL FARM	30.5	15.9	6.3	10.7*	3.0
Cost ($/ha)		$13.13	$42.40	$29.56	$93.01

* Although soil erosion was only restricted to 15 MT/ha, practices required to reduce the Tama and Dinsdale soils to this level caused erosion levels to be considerably less on the Muscatine and Garwin soils.

a farm plan designed to reduce sediment delivery by an equivalent amount (to 5.0 MT/ha) costs only $22.14/ha or 75% of the cost of the soil erosion plan. Likewise reducing soil erosion by 90% reduces sediment delivery to 1.1·MT/ha and costs $93/ha. This level of sediment delivery can be achieved through a sediment delivery plan at a cost of only $48 or about 53% of the cost of the soil erosion plan.

Texas Example Farm

The example farm for Texas is cash crop-hay operation consisting of 100 ha of tillable land plus an unspecified but fixed amount of additional permanent pasture ground assumed to be unsuitable for crops. Since the market for hay is fairly limited in this area of Texas, the area of hay was fixed for all plans at about 19% of total crop acres (19 ha). Other crops considered in this farm were cotton, wheat, and sorghum.

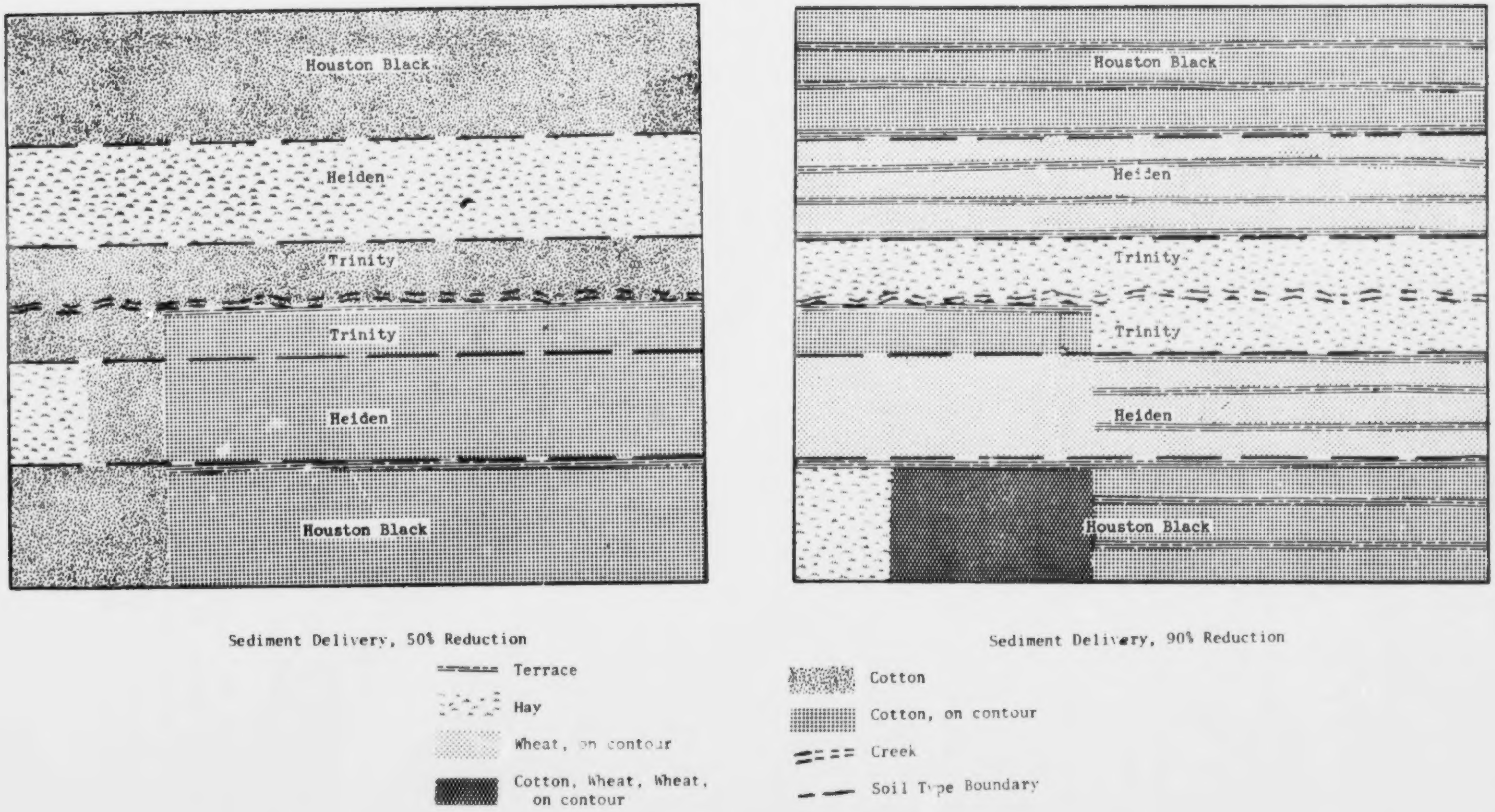
Methods for maintaining cotton yields under either conservation tillage or no-tillage have not been fully developed. Also, the soils in this area are predominately fine-textured clays, which decreases their suitability for these reduced tillage practices. Thus neither conservation tillage nor no-tillage was considered as an option for this farm.

Although no acreage restrictions exist for the crops grown in this area, cotton, wheat, and sorghum have a base acreage which is established by ASCS for each farm. If a farm exceeds the base, it is not eligible for price supports or government commodity loans. Since many farmers participate in the program significant areas of sorghum and wheat are grown even though cotton provides the highest net return per hectare. These base allotments vary from farm to farm and were not considered on the example farm. Although this has the effect of distorting the actual levels of income, soil erosion, and sediment delivery, it does not have a large effect on the evaluation of SWCPs, particularly terraces, for the example farm.

In this area of Texas, the topography is typically complex with few long slope lengths. The total slope length for all three soils combined is only 250 m, thus the total farm width is 500 m. Therefore the 100 ha example farm is 2,000 m long. Since these dimensions are not conducive to illustrating farm plans, the plans are not drawn to scale, rather the farm is shown with a proportionately expanded width.

Under the base system, 81 ha of cotton and 19 ha of hay are grown, with all the hay being produced on the Trinity soil. Although this soil has the lowest slope gradient (0.5%), the other soils are relatively more efficient in producing cotton than hay.

A 50% reduction in sediment delivery is obtained by using a two terrace system which covers about 40% of the farm (Figure 8-3). One terrace is next to the stream, the other approximately halfway up the slope at the lower edge of the Houston-Black soil. All of the terraced area is in contoured cotton. Hay production is moved to the Heiden soil, the most erosive area of the farm.


Houston Black
Heiden
Trinity
Trinity
Heiden
Houston Black
Houston Black
Heiden
Trinity
Trinity
Heiden
Houston Black
Terrace
Hay
Wheat, on contour
Cotton, Wheat, Wheat, on contour
Cotton
Cotton, on contour
Creek
Soil Type Boundary

Sediment Delivery, 50% Reduction

Sediment Delivery, 90% Reduction

FIGURE 8-3. FARM PLANS FOR REDUCING SEDIMENT DELIVERY, TEXAS.

This simple two-terrace system was not adequate to obtain a 90% reduction in sediment delivery, thus a system with shorter terrace intervals is required, having 12 terraces across the farm, six on each side of the stream (Figure 8-3). None of the terrace options available for this farm had both reduced terrace intervals and a terrace along the streambank. Therefore to obtain the 90% reduction in sediment delivery, hay production was shifted to the Trinity soil to minimize soil erosion in the area having the highest SDR. Even with the more intense terrace system, cotton production is severely cut back with this plan. Nearly 40 ha of wheat are grown replacing acreage which was in cotton on both the base plan and the 50% reduction plan.

A 50% reduction in soil erosion is obtained by a single terrace system over nearly half the farm. As in the sediment delivery plan, hay is shifted to the Heiden soil at the 50% reduction level. The additional cost of this plan over the plan for reducing sediment delivery by 50% is primarily due to contouring on all the Houston-Black soil.

To achieve a 90% reduction in soil erosion, a terrace system covers about 90 ha , with the remaining 10 ha removed from crop production. The total cotton area is reduced to about 10 ha. These latter two factors account for the substantially greater cost of this plan over all the other plans for the Texas farm.

TABLE 8-13. TEXAS: FARM PLANS FOR REDUCING SEDIMENT DELIVERY AND SOIL EROSION

		Farm Plans For:			
	Base Plan	50% Reduction in Sediment Delivery	90% Reduction in Sediment Delivery	50% Reduction in Soil Erosion	90% Reduction in Soil Erosion
Sediment Delivery (MT/ha)					
Houston-Black	15.2	11.1	3.0	7.5	1.0
Heiden	40.5	5.4	1.1	11.4	1.0
Trinity	4.2	20.6	1.2	24.0	5.0
TOTAL FARM	21.8	11.0	2.0	12.2	1.8
Soil Erosion (MT/ha)					
Houston-Black	40.6	34.1	15.2	24.0	5.0
Heiden	83.5	19.8	6.7	24.0	5.0
Trinity	4.2	25.4	3.6	24.0	5.0
TOTAL FARM	48.3	27.4	9.9	24.0	5.0
Cost ($/ha)	----	$10.19	$52.81	$13.34	$120.02

Further Discussion of Farm Plans on the Example Farms

Thus far, primary exmphasis has been on the types of differences that occur between plans oriented towards reducing sediment delivery versus those oriented towards reducing soil erosion. Some practices are equally effective for both types of farm plans. In particular, the four practices listed in the previous section as having the highest potential levels of cost-effectiveness occur in the farm plans for each farm on which they were considered applicable. Contouring was the most extensively used practice occuring on all plans on all the farms. Conservation tillage was not applicable in Texas, but was the primary tillage practice in both Iowa and New York. No-tillage was only considered on the New York farm and was the primary practice for growing corn grain on all the farm plans except the plan for reducing sediment loss by 50%. Strip cropping was also used in all farm plans where sod-based rotations were part of the cropping system (only continuous hay was considered in Texas).

On the other hand, sod-based rotations were shown to have a fairly low level of cost-effectiveness. On the New York farm, a certain amount of hay was required to feed the dairy herd, and was part of the base system. However, no significant increase in hay area occurred for any of the farm plans investigated. In Iowa, small areas of rotation hay occurred at the 90% reduction level. However, this was one of the major factors in creating the greatly increased costs at this level of reduction. No variance in hay area was allowed on the Texas farm, thus this practice was not applicable.

In general, the differences in the types of practices used on the three farms is more dependent on the particular soils and type of farming operation than on the area of the country. For example, strip cropping was used only on the New York example farm. However, farms with cattle enterprises in Texas, Iowa, or other areas would normally grow hay crops, and could readily employ strip cropping. Likewise, conservation tillage could be used in many areas of Texas, even though it was not used on the Texas example farm. Thus, evaluation of practices generally must be done on an individual farm basis rather than on a regional basis.

Conservation Tillage versus Conventional Tillage on the Example Farms

As previously noted, conservation tillage was assumed to be the normal tillage practice on the Iowa and New York example farms. These practices are compared for the base plan on these two farms in Table 8-14.

For both farms, income (above selected variable costs) is reduced by about $5/ha when moldboard plowing is used instead of chisel plowing. In both cases, the two tillage systems were assumed to produce equal yields, thus the difference in income is due only to differences in field time and machinery maintainence costs.

Levels of soil erosion and sediment delivery with conventional tillage on the Iowa farm are nearly double the levels with conservation tillage. In New York, however, conventional tillage increases soil erosion by only about 30% and sediment delivery by about 16%. With the base plan in New York, all the Lima-Kendaia soil is in continuous corn silage. There is little difference (< 4%) in levels of soil erosion between conservation and conventional tillage on crops where crop residues are removed, such as corn silage. Thus the switch in tillage practices has a small effect on soil erosion levels. Since the Lima-Kendaia soil has the higher SDR of the two soils on the farm, the differences in sediment delivery are even less than the differences in soil erosion.

Cost Associated with Reducing Sediment Delivery

There are several methods of looking at costs of soil and water conservation practices. The average cost per hectare is a useful method of measuring the potential effect on farm income. This is the cost shown in Tables 8-11, 8-12, and 8-13. However, average cost per hectare does not necessarily relate costs to a particular goal. If the goal is to reduce the amount of sediment reaching a stream, then a more meaningful cost is the cost per unit of reduction in sediment delivery. This would be particularly true if it were also possible to quantify the benefits of reducing sediment delivery. In theory, the optimal level of reduction in sediment delivery occurs where the marginal cost of reducing sediment delivery equals the marginal benefit. At any given level of reduction in sediment delivery, marginal cost is defined as the additional cost of keeping one additional metric ton of sediment out of a waterway, while marginal benefit is the additional benefit which would be received by restraining that one additional ton. No attempt was made in this study to estimate water quality benefits

TABLE 8-14. CONSERVATION TILLAGE* VERSUS CONVENTIONAL TILLAGE* ON THE IOWA AND NEW YORK EXAMPLE FARMS

Base Farm Plan	Income above Selected Variable costs ** $/ha	Soil Erosion MT/ha	Sediment Delivery MT/ha
Iowa:			
Conservation	705	30.5	15.5
Conventional	699	61.8	31.1
New York:			
Conservation	1083	38.5	26.1
Conventional	1078	50.3	30.4

* Conservation tillage is chisel plowing with one secondary tillage operation. Conventional tillage is moldboard plowing followed by two secondary tillage operations.

** Income is total farm income and thus includes income from the hog enterprise in Iowa and the dairy enterprise in New York.

of reduced sediment delivery to streams. One should keep in mind that while marginal costs of reducing sediment delivery increase as sediment delivery is restricted, marginal benefits are likely concurrently to decrease.

For each of the example farms, marginal cost and cost-effectiveness curves were plotted. The curves on all the farms had a common pattern. At low levels of reduction in sediment delivery, cost-effectiveness values were high, and up to a point, dropped rather slowly as levels of reduction increased. On each farm, however, there was a point where cost-effectivness began to drop sharply. This occurred between the 70% and 90% levels of reduction. The marginal cost curve had a reverse shape, starting low, increasing gradually at first up to a certain point where it suddenly rose rapidly. Marginal costs also tend to rise in a step-wise manner. As sediment delivery is reduced, marginal cost increases in jumps with fairly level areas in between. Comparing these points of rapid increase with the farm plans, it was seen that they occurred with the introduction of major new practices. After a new practice is introduced on part of the farm, marginal cost remains nearly level until that practice is used on the whole farm, then a rapid jump occurs with the introduction of a new more intensive practice. This pattern can be clearly seen in the cost curves for Texas, shown in Figure 8-4. The marginal cost curve is at first nearly level with a slight jump at a 55% reduction. It rises rapidly above the 75% reduction level. Reductions below the 75% level are achieved with non-structural practices, including contouring and crop placement combined with a simple terrace system. This system is shown in the farm plan for a 50% reduction in Figure 8-3. At the 75% reduction level, the whole farm is in this simple terrace system; thus to achieve greater reductions, new, less cost-effective practices must be introduced. In this case three things happen together which have the combined effect of rapidly increasing marginal cost. They are introduction of a more intense terrace system, substituting wheat for cotton, and taking some land out of production.

The existence of these points where marginal costs suddenly rise rapidly is partly a function of the number and type of practices available on a farm. For the New York farm, for example, which had more options available, the point of rapid rise occurred at a higher level (about 85%) and was somewhat less abrupt. It is clear that to the extent that critical point exists for a farm, maximum reductions per unit of cost will occur if levels of reduction are below this point. Thus one general conclusion is that it is more cost-effective to achieve moderate reductions in sediment delivery over several farms than to attempt large reductions on only a few farms, providing all farms have roughly equivalent initial levels of sediment delivery.

The Relationship between Soil Erosion and Conservation

The farm plans for controlling either soil erosion or sediment delivery discussed earlier did not take into account the long run impact of reduced soil loss on crop yields. Estimation of such long run yield effects has been attempted by some researchers by relating yields to topsoil depth (e.g., Seitz et al 1978). Future topsoil depth has been estimated by substracting erosion calculated from the USLE from initial topsoil. This probably overestimates the reduction in average topsoil depth in a field because no adjustment is made for topsoil redeposited either within the same field or on a field between the field in question and the stream. The level of knowledge in this area is inadequate to make definite statements about the effects of redeposition on topsoil depth and future crop yields. As stated earlier, there are no generally accepted methods of calculating sediment delivery ratios. There is less agreement on where the sediment not delivered to stream is redeposited. This section attempts to shed some light on the subject by use of examples based on the New York, Iowa and Texas farms.

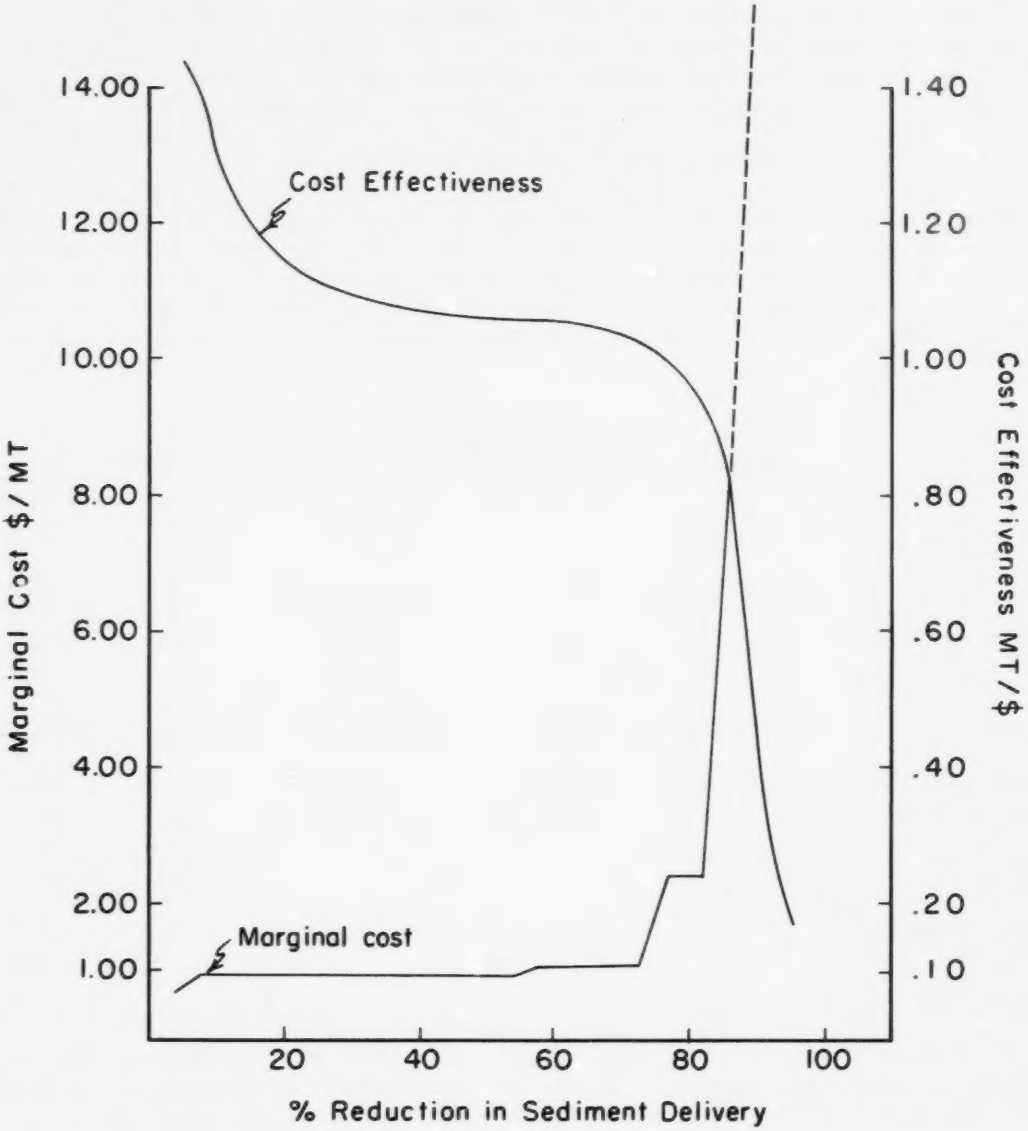


FIGURE 8-4. COST-EFFECTIVENESS AND MARGINAL COST FOR REDUCING SEDIMENT DELIVERY ON THE TEXAS EXAMPLE FARM

Soil erosion, as defined in Section 4, is the detachment and transport of soil by water. Soil erosion levels given for the farm plans for the example farms (Tables 8-11, 8-12, and 8-13) represent the expected total soil erosion on those farms over a year as calculated from the USLE. When, as is true for most fields, the SDR is less than 1, soil is redeposited and at least part of this redeposition is within cropped fields. This could be in depressions within the field or on fields over which runoff flows before it reaches the stream.

On the example farms, all slopes were assumed to be convex on those soil types away from the stream and concave on the soils adjacent to the stream. If it is assumed that all redeposition occurs on the concave areas, then net soil loss from these areas is considerably less than the soil erosion levels shown in Tables 8-11, 8-12, and 8-13. In Table 8-15, net soil loss for each soil type is calculated for the base plan for each example farm. This calculation is based on the assumption that all eroded soil which does not leave the farm as sediment delivery is redeposited on the soil type which lies next to the stream. Thus net soil loss from the farm is the same as the level of sediment delivery. This would not be true, however, if terraces or other types of sediment traps are on the farm, since, in this case, some of the redeposition of sediment occurs in these structures instead of on the lower fields.

TABLE 8-15. SOIL EROSION AND NET SOIL LOSS
FROM THE EXAMPLE FARMS UNDER THE BASE PLANS (MT/ha)

New York					
	Honeoye-Lima		Lima-Kendaia		Total Farm
Soil Erosion	16.9		60.2		38.5
Net Soil Loss	16.9		25.3		26.1
Iowa	Muscatine	Tama	Dinsdale	Garwin	Total Farm
Soil Erosion	5.0	20.8	128.9	9.1	30.5
Net Soil Loss	5.0	19.6	68.3	-27.2	15.5
Texas	Houston-Black	Heiden		Trinity	Total Farm
Soil Erosion	40.6	83.5		4.2	48.3
Net Soil Loss	40.6	83.5		-128.5	21.8

Net soil loss is considerably less than USLE soil erosion on the Lima-Kendaia, Garwin, and Trinity soils. On the latter two, redeposition actually exceeds the level of soil erosion, thus net soil loss is negative.

The net soil loss levels shown in Table 8-15 are based on the calculated SDRs and thus may not be precise. However, they do show that levels of gross soil erosion, particularly on complex slopes, are not necessarily the best indicator for the conservation of topsoil and thus for estimating long run yield effects of reducing soil erosion.

Cost-Sharing

Cost-sharing with farmers is one method of encouraging adoption of SWCPs. The Agricultural Stabilization and Conservation Service (ASCS) has a continuing program of cost-sharing with farmers for adoption of practices principally designed for preventing soil erosion. Only practices which meet fairly exact standards of both ASCS and the Soil Conservation Service (SCS) are cost-shared. Recently, money has been authorized for cost-sharing with farmers on practices specifically aimed at reducing non-point source pollution from agricultural land.[1] Since there can be a significant difference in the effectiveness of SWCPs for reducing sediment load compared to the effectiveness for reducing soil erosion, it would be expected that the structure of the new cost-sharing program would be somewhat different from the traditional ASCS program.

The goal of any cost-sharing program should be to encourage farmers to meet the program's objectives at the least cost to both the farmer and the cost-sharing agency. If cost-sharing is not available on the least costly system of practices or it is not proportional to the actual costs of practices to the farmers, farmers may adopt practices which are more costly and/or less effective for reaching the water-quality goals. However, because of monitoring problems and variations between farms, it may be difficult to cost-share on certain practices, particularly those which are non-structural, require annual on-site checks to ensure compliance, and/or do not have specific costs associated with their implementation. Practices such as contouring, minimum tillage, no-tillage, and crop rotations would generally fall into this category. Also, for practices which influence farm costs primarily by their effect on yield (primarily no-till and conservation tillage), it may not be possible to determine a level of cost-sharing which is actually proportional to farm costs, since these can vary from field to field, and in some cases, are insignificant or even negative.

All of the farm plans presented in the previous section were derived assuming no cost-sharing funds were available. However, cost-sharing funds generally are available for many practices through the Agricultural Stabilization and Conservation Service (ASCS). Practices which are eligible for cost-sharing must meet the criteria of both the ASCS and SCS. To determine how present cost-sharing programs affect farm plans for controlling sediment delivery, the LP model for each example farm was rerun with current cost share levels included for eligible practices. Levels of cost-sharing were taken from current ASCS programs for a representative county in each state. The counties were Tompkins County, New York; Grundy County, Iowa; and Collin County, Texas.

Cost-sharing under the ASCS program generally falls into three categories: a fixed percentage of initial installation costs for structures, a fixed percentage of establishment costs for sod crops, and a single per hectare payment for other eligible practices such as strip cropping, minimum tillage and/or contouring. Eligible practices in the third category vary significantly from area to area and from year to year, and with the possible exception of strip cropping, are not available at all in most areas. Cost-sharing rates for the three counties are given in Table 8-16.

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[1] Section 35. Clean Water Act of 1977.

Although cost-sharing is currently available at 90% for structural practices in Tompkins County, New York, for most years and for most counties in the state, rates of 60%-75% would be more typical. Still, rates for New York are higher than for the other areas.

For both Texas and Iowa, availability of cost-sharing funds had no significant effect on practices used in farm plans for sediment reduction. The terrace system with conventional (SCS) spacing was assumed to be the only system eligible for cost-sharing. For both Texas and Iowa, this system had a low level of cost-effectiveness and did not occur in farm plans except at extremely high levels of restraint. Although cost-sharing caused a slight increase in the areas with this terrace system, it had no significant overall effect.

Cost-sharing could potentially have brought significantly more hectares of hay into the Texas farm plans, if hay had not been restrained to a maximum of 19% of total area. In Iowa, however, adding hay area had a low cost-effectiveness, which was not significantly affected by cost-sharing on hay

TABLE 8-16. COST-SHARING RATES FOR
ELIGIBLE SWCPS ON THE EXAMPLE FARM

	New York	Iowa	Texas
Terrace and Diversion Systems[*,#]	90%	60%	50%
Conservation Tillage[&]	--	$12	--
Strip Cropping[&]	$75	$10	--
Permanent Sod Cover[¢]	75%	60%	50%

[*] Percent of initial construction cost

[#] Cost-sharing for terraces available only for terrace intervals meeting SCS specifications

[&] One-time per hectare payment for initial implementation

[¢] Percent of establishment cost

establishment. Even though cost-sharing did not affect optimal farm plans on these two example farms, there was no significant amount of cost-sharing money available for practices in the farm plans with less than about an 80% reduction in sediment delivery. Thus the current cost-sharing program would not offer a significant incentive to farmers where only moderate decreases in sediment delivery were desired.

For the New York example farm, cost-sharing played a more significant role. More practices are eligible for cost-sharing, and cost-sharing rates are higher. Thus availability of cost-sharing money changed the optimal farm plans at all levels of reduction, primarily by causing substitutions of more extensive terraces and diversions for contouring and no-till to occur. Although cost-sharing results in lower farm costs, the total cost is increased. Table 8-17 shows that total cost, which includes actual farm cost plus cost-sharing funds, nearly doubles when cost-sharing is available at the 90% reduction level and increases by about 20% at the 50% level.

The cost increases with cost-sharing are due primarily to a lack of funding for certain practices which may be highly cost-effective. These would include practices such as conservation tillage, no-till, contouring, and terrace systems with wider intervals than provided for by current specifications. Providing cost-sharing for non-structural practices may require some sort of annual monitoring. This would certainly have the effect of increasing costs for the responsible agency which would partially offset the benefits of these practices. Further analysis in this area would be required to fully evaluate these alternatives.

Technical evaluation of the feasibility of the wider terrace intervals used on the example farms was not attempted. Thus in some cases, these terrace systems may not be possible since runoff and/or sedimentation may exceed their capacity. This would be particularly true if continuous highly erosive, row crops are grown. However, the possibility of combining wider terrace spacings with other practices such as conservation tillage, no-till or stripcropping would certainly be feasible under many conditions and would often be more cost-effective than a more intense terrace system.

TABLE 8-17. EFFECT OF COST-SHARING ON COSTS
FOR REDUCING SEDIMENT DELIVERY, NEW YORK ($/ha)

	% Reduction			
	50%		90%	
	with Cost-sharing	without Cost-sharing	with Cost-sharing	without Cost-sharing
Farm Cost	1.00	5.10	19.70	27.40
Cost-share	5.20	----	32.40	----
Total Cost	6.20	5.10	52.10	27.40

Potential Effects of Farm Plans for Reducing Sediment Delivery on Losses of Other Pollutants

Farm plans for reducing sediment delivery may not have similar effects on the losses of other potential pollutants. In this section, results from Sections 4, 6, and 7 are used to evaluate some possible effects of farm plans for reducing sediment delivery on the losses of nutrients and pesticides. First, results from Section 4 are used to describe the types of expected changes that would occur with the farm plans. Then the CNS model is used to predict nutrient losses for the Iowa example farm. Finally the CPM model is used to predict losses of atrazine and several alternative herbicides for continuous corn with several SWCPs on the Iowa farm.

Tables 4-1 and 4-2 list pollutants in categories according to their degree of adsorption to soil particles. Table 4-4 lists individual SWCPs as "Effective", "Slightly-effective", or "Not effective" in controlling pollutants in each of the adsorption categories. Table 4-4 is used as the basis for much of the following discussion. "Strongly adsorbed" substances move primarily with sediment and are concentrated on the fine particles. Thus in general, the effectiveness of a practice for controlling strongly adsorbed substances will be highly correlated to its effectiveness in controlling sediment delivery. However, practices, such as terraces, which achieve their effect primarily by providing for redeposition of eroded sediment will be relatively less effective than practices which are effective due to reducing splash energy or increasing infiltration. Thus terraces will reduce losses proportionately less than they reduce sediment delivery. The non-structural practices, however, generally increase surface cover and thus reduce splash erosion. For these practices, particularly no-till, conservation tillage, and sod-based rotations, reductions in strongly adsorbed substances should be nearly proportional to reductions in sediment delivery. In general, farm plans designed to reduce sediment delivery should be effective in reducing losses of strongly adsorbed substances, but this reduction will usually be less than proportional to the reduction in sediment delivery.

"Moderately adsorbed" substances move primarily with overland flow, thus the effectiveness of SWCPs in reducing losses depends primarily on reducing surface runoff. Sod-based rotations is the only practice evaluated in this chapter which is listed as effective in Table 4-4. However, sod-based reductions were found to generally have a low level of cost-effectiveness for reducing sediment delivery and did not occur (as an increase over the base system) extensively in any of the farm plans. Contour farming, conservation tillage, no-tillage, and terrace did occur commonly in the farm plans and were found to have potentially high levels of cost-effectiveness. These four practices are listed as slightly effective in controlling moderately adsorbed substances. Thus farm plans for reducing sediment delivery should reduce levels of moderately adsorbed substances, but probably considerably less than the reductions in sediment delivery.

Only contouring and sod-based rotations are listed as slightly effective in controlling non-adsorbed pollutants, all other practices evaluated in this chapter are listed as "Not-effective." Thus it would not be expected that farm plans for reducing sediment delivery would have any significant effect on losses of these substances.

Levels of Nutrient Loss on the Iowa Example Farm

Results from the CNS model (Section 6) were used to estimate losses of nutrients from the Iowa farm under the base plan and with the farm plans for achieving a 50% and a 90% reduction in sediment delivery. These results are presented in Tables 8-18, 8-19, and 8-20.

Solid-phase Nutrients. The change in losses of solid-phase nutrients as sediment delivery is reduced depends on both the level of sediment delivery and the enrichment ratio (Section 6). The CNS model assumes the enrichment is constant and independent of the level of soil erosion or sediment delivery.

TABLE 8-18. LOSSES[*] OF SOLID-PHASE NUTRIENTS FOR THE IOWA EXAMPLE FARM

	Base Plan	50% Reduction in Sediment Delivery	90% Reduction in Sediment Delivery
Constant Enrichment Ratio[#]			
Enrichment Ratio	1.25	1.25	1.25
Nitrogen	9.7	4.8	.97
Phosphorous	13.2	6.6	1.3
% Reduction from Base Plan	--	50%	90%
Changing Enrichment Ratio[&]			
Enrichment Ratio	1.25	1.5	2.25
Nitrogen	9.7	5.8	1.7
Phosphorous	13.2	7.7	2.4
% Reduction from Base Plan	--	40%	82%

[*] Average annual losses in kg/ha.

[#] The losses are based on results from the CNS model (Section 6). However, this model assumes a constant enrichment ratio of 2.0. Results were adjusted for an enrichment ratio of 1.25 for the sake of comparison.

[&] From Casler and Jacobs (1975).

TABLE 8-19. AVERAGE ANNUAL LOSSES OF DISSOLVED NUTRIENTS WITH TWO CROP ROTATIONS AND SELECTED SWCPS FROM THE IOWA MUSCATINE SOIL

	Soil Erosion (MT/ha)	Dissolved N in Runoff (kg/ha)	Dissolved P in Runoff (kg/ha)	Leached Nitrogen (kg/ha)
Corn-Soybeans Rotation				
Conservation tillage	7.2	4.7	0.17	31.0
plus Contouring	3.3	3.7	0.13	32.2
plus Terracing	2.0	3.2	0.10	33.2
Continuous Corn				
Conservation tillage	5.0	11.0	0.16	65.3
plus Contouring	2.3	8.8	0.13	68.1
plus Terracing	1.5	7.2	0.11	69.8

If this is true, reductions in losses of solid-phase nutrients will be proportional to reductions in sediment delivery. However, if the enrichment ratio increases as levels of sediment delivery decrease, then reductions in losses of solid-phase nutrients will be proportionately less than reductions in sediment delivery. There appears to be considerable debate on the relationship between the enrichment ratio and levels of soil erosion. This relationship may depend on both the properties of the soil and the types of SWCPs implemented, and at present, is not known. Casler and Jacobs (1975) developed a curve relating the enrichment ratio for phosphorous with the level of soil erosion. This curve was based on research reported by Massey and Jackson, (1952). Table 8-18 shows levels of losses of solid-phase nutrients for the Iowa farm using both the constant enrichment ratio as in the CNS model, and the variable enrichment ratio, developed by Casler and Jacobs.

It appears that reducing sediment delivery will definitely reduce losses of solid-phase nutrients. However, the amount of reduction depends on the presently unknown relationship between the enrichment ratio and the levels of soil erosion and sediment delivery.

Dissolved Nutrients. Losses of dissolved nutrients depend, among other things on both the quantity of surface runoff water and the amounts of fertilizer applied. Thus effects of individual practices, where rates of fertilization do not change may not be the same as the effects of implementing farm plans which change the crop mix and thus rates of fertilizer application.

Since soybeans require no nitrogen fertilizer, twice as much nitrogen fertilizer is applied with continuous corn as with a corn-soybeans rotation. Using the Muscatine soil from the Iowa example farm, losses of soluble nutrients for several SWCPs with continuous corn and with a corn-soybeans rotation are given in Table 8-19.

Total dissolved nitrogen losses with continuous corn are roughly double the losses with corn-soybeans while losses of dissolved phosphorous are about the same under either cropping system. Losses of both nitrogen and phosphorous in runoff are decreased by both contouring and terracing, but the reductions are proportionately less than the reductions in soil erosion. Also levels of leached nitrogen increase as dissolved nitrogen losses decrease such that there is little change in the total dissolved nitrogen leaving the field.

Table 8-20 shows the edge-of-field losses of dissolved nutrients for the Iowa farm plans for reducing sediment delivery. It should be noted that these are edge-of-field losses and not estimates of stream loading. The transport of these dissolved substances was not investigated. Note also that dissolved nutrient losses are not reduced by the same percentage as the reductions in sediment delivery.

The combinations of practices used in the Iowa farm plans tend to slightly increase nitrogen losses and to decrease losses of phosphorous by small amounts. The magnitude of the changes, however, is within the expected accuracy of the CNS model, thus the changes are not considered to be significant. In particular, the model does not include the effects of phosphorous leaching from crop residues on losses of soluble phosphorous. As levels of sediment delivery are decreased on the Iowa farm, there is a shift from predominately fall tillage to predominately spring tillage. Thus more crop

TABLE 8-20. EDGE-OF-FIELD LOSSES* OF DISSOLVED NUTRIENTS FOR THE IOWA EXAMPLE FARM

	FARM PLANS		
	Base Plan	50% Reduction in Sediment Delivery	90% Reduction in Sediment Delivery
Dissolved Inorganic Nitrogen in Runoff	6.9	6.5	7.2
Dissolved Phosphorous in Runoff	0.20	0.18	0.15
Leached Inorganic Nitrogen	37.7	39.4	41.1

* Average annual losses kg/ha

residues are left on the field over the winter and early spring months as levels of sediment delivery are reduced. This may have the potential for raising the losses of dissolved phosphorous.

The increase in levels of nitrogen losses as sediment delivery is reduced is primarily due to an increase in corn grain acreage relative to soybean acreage. For the situation of the Iowa example farm, the change in levels of dissolved nitrogen losses (runoff and leached) are small since the change in corn acreage is fairly small (about 10%). However, if there were a large shift from soybeans to corn on a farm, losses of dissolved nitrogen could increase significantly.

Evaluation of Alternative Methods for Reducing Herbicide Losses from Continuous Corn

In Section 7, the effect of individual SWCPs on losses of pesticides was investigated using the CPM model. One of the conclusions from that section was that sizeable reductions in losses of pesticides in runoff may often be achieved with SWCPs. However, in situations where short-lived pesticides may be substituted for more persistent pesticides, losses in runoff may also be reduced, and in some situations, these reductions will be greater than these achieved with SWCPs. In Section 9, other methods of reducing pesticide losses from cropland are discussed. These include use of alternative application methods, use of alternative pesticides, and integrated pest management practices.

Only a few of the techniques mentioned in Sections 7 and 9 may be applicable in any given cropping situation. For example, few integrated pest management practices are currently available for reducing herbicide use on corn. However, other methods of reducing losses of herbicides do exist. These include SWCPs, changing the type of herbicide used, and alternative application methods.

Many different herbicides and combinations are currently available for use on corn. Currently the single most common method of weed control in Iowa is a pre-emergence surface-spray application of atrazine and alachlor, usually in combination with at least one cultivation (Richard Fawcett, Personal Communication, 1978). This is the herbicide combination which was used for determining costs on the Iowa example farm.

Atrazine, the most widely used herbicide is more persistent than many other herbicides and is commonly found as a contaminant in surface waters in corn growing areas (Morley, 1977; Richard et al. 1975). Thus decreasing herbicide losses in runoff could potentially have water quality benefits in these areas.

In this section, several techniques for reducing herbicide losses in runoff are investigated. These include use of alternative herbicides (cyanazine plus alachlor and butylate plus cyanazine instead of atrazine plus alachlor), alternative application methods (surface applied versus incorporated) and SWCPs (conservation tillage, contouring, and terracing). Properties and application methods for the three herbicide combinations are given in Table 8-21. Note that the cyanazine plus alachlor (Cy-al) combination has the same application procedure as atrazine plus alachlor (at-al) but cyanazine is less persistent in soil than atrazine. Butylate plus cyanazine (bu-cy) has the same persistence as cy-al but is applied earlier and incorporated. It should also be noted that since incorporation requires thorough mixing of the top 15 cm of soil (two secondary tillage operations after application), it is not compatible with either conservation tillage or no-tillage.

TABLE 8-21. PROPERTIES AND TYPICAL APPLICATION METHODS FOR THREE COMMON CORN HERBICIDE COMBINATIONS

	Atrazine plus Alachlor (at-al)	Cyanazine plus Alachlor (cy-al)	Butylate plus Cyanazine (bu-cy)
Common Trade Names	Aatrex plus Lasso	Bladex plus Lasso	Sutan+ plus Bladex
Estimated Half-life in soil (days)	50*	14	14
Adsorption Group (Table 4.3)	B	B	B
Application Rate Kg/ha a.i.#	1.25 2.5	1.25 2.5	4.0 1.25
Application Method	Surface Spray	Surface Spray	Incorporation
Application Time	Pre-Emergence (May 15)	Pre-Emergence (May 15)	Pre-Plant (May 1)
Cost $/ha	$30.00	$32.00	$45.00
Available tillage practices	All	All	Conventional only

* Estimated half-life for atrazine only. Half-life for alachlor is 14 days.

# Application rates are in kilograms of active ingredients (a.i.). Upper number refers to rate for first herbicide listed in each combination.

The base condition shown in Table 8-22 is straight-row conservation tillage using atrazine plus alachlor. As discussed earlier, conservation tillage is the most economical tillage system for growing corn in this area of Iowa. Under conventional tillage, herbicide losses are over 25% greater. The two SWCPs evaluated, contouring and contouring plus terracing, reduce herbicide losses by 43% and 72%, respectively, when the herbicide is not changed. Switching from the at-al combination to the cy-al combination achieves only a fairly small (12%) reduction in herbicide loss. This reduction is due to the reduced half-life of cyanazine compared to atrazine (Table 8-21).

The largest reduction is brought about by switching to the incorporated bu-cy combination. Although not evaluated here, even larger reductions could be achieved with contouring and terracing. However, incorporation

TABLE 8-22. ESTIMATES OF HERBICIDE LOSSES IN RUNOFF FROM CONTINUOUS CORN ON THE TAMA SOIL FROM THE IOWA EXAMPLE FARM

	Average Annual Loss g/ha-yr	Effectiveness* g/ha-yr	% Reduction	Level of Soil Erosion MT /ha-yr
Base Condition #	22.1	--	--	15.0
Atrazine-Alachlor (at-al)				
SR-Conv &	27.9	*	*	37.1
SR-Cons	22.1	(¢)	(¢)	15.0
Cont-Cons	12.7	9.4	43	7.5
Terr-Cons	6.2	16.1	72	5.3
Cyanazine-Alachlor (cy-al)				
SR-Conv &	24.7	*	*	37.1
SR-Cons	19.5	2.6	12	15.0
Cont-Cons	11.2	10.9	49	7.5
Terr-Cons	5.4	16.7	76	5.3
Butylate-Cyanazine (bu-cy)				
SR-Conv &	4.3	17.8	81	37.1

* Effectiveness is the reduction in losses from the base condition.

The base condition is the least-cost method of growing corn. In this case it is straight-row conservation tillage using the atrazine plus alachlor herbicide combination.

& Abreviations for SWCPs are: SR - straight-row; Conv - conventional tillage; Cons - conservation tillage; Cont - contouring; Terr - contouring plus terracing.

¢ Base Condition.

requires conventional tillage and thus brings about a significant increase in the level of soil erosion. Thus in areas where conservation tillage is a feasible soil conservation practice, incorporation of herbicides as a means for controlling losses of herbicides may require implementation of alternative soil erosion control practices which would be less cost-effective than conservation tillage. However, in areas where conservation tillage is not economically feasible, such as the area of the Texas example farm, incorporation may be an effective means of reducing herbicide losses.

Costs for these alternative practices are shown in Table 8-23 as the level of cost increase expected above the cost of the base condition. Cost-

TABLE 8-23. COST AND COST-EFFECTIVENESS OF VARIOUS PRACTICES FOR CONTROL OF HERBICIDE LOSSES FROM THE IOWA TAMA SOIL

	Costs* $/ha Tillage Costs	SWCP Cost	Herbicide Cost	Total Cost	Cost-Effectiveness[#] g/$
Atrazine-Alachlor (at-al)					
SR-Conv [&]	5.00	--	--	5.00	--
SR-Cons	(¢)	(¢)	(¢)	(¢)	(¢)
Cont-Cons	--	2.22	--	2.22	4.2
Terr-Cons	--	66.67	--	66.67	0.2
Cyanazine-Alachlor (cy-al)					
SR-Conv	5.00	--	2.00	7.00	--
SR-Cons	--	--	2.00	2.00	1.3
Cont-Cons	--	2.22	2.00	4.22	2.6
Terr-Cons	--	66.67	2.00	68.67	0.2
Butylate-Cyanazine (bu-cy)					
SR-Conv	9.00[**]	--	15.00	24.00	0.7

* Costs represent the increase in costs from the base condition, which is straight-row conservation tillage using atrazine plus alachlor.

[#] Cost-effectiveness is the level of effectiveness, from Table 8-22 divided by the total cost.

[&] Abbreviations are listed in Table 8-21.

[¢] Base condition.

[**] Incorporation requires one additional secondary tillage operation, thus increases the tillage cost above that for conventional tillage alone.

effectiveness, which is the reduction in grams of annual pesticide loss per dollar of cost is also tabulated. It should be remembered that these costs can vary considerably, particularly the cost for contouring and the relative prices of the herbicides. Thus the cost-effectiveness values are only indicative and will vary from field to field and from year to year.

Given the cost assumption for contouring used here, it is the single most cost-effective practice for reducing herbicide loss. Contouring and switching to the cy-al combination results in a reduction of nearly 50% in herbicide loss at a fairly small cost per hectare. However, to achieve greater reductions either incorporation or a terrace system must be implemented at a significantly higher level of cost and reduced cost-effectiveness.

If any practice is to be implemented solely for the purpose of reducing pesticide losses in runoff, the implications of these cost-effectiveness values should be considered. The maximum value in Table 8-23 is 4.2 for contouring. Thus by implementing contouring, 4.2 grams of herbicide are prevented from leaving the field per dollar spent. To achieve an 80% reduction (switching to the incorporated bu-cy combination) only 0.7 grams are retained per dollar spent. These values are based on edge-of-field loss and not stream loading, thus the potential exists for degradation and deposition (adsorption to deposited soil) before these chemicals reach a waterway. Thus cost-effectiveness values for these alternatives may well be less than one gram per dollar for many situations. Therefore it would seem that careful consideration of benefits should be undertaken before implementing on-field practices for reducing pesticide losses in runoff.

CONCLUSIONS

1.) Since sediment delivery is the product of soil erosion times the sediment delivery ratio (SDR), SWCPs may affect sediment delivery by either changing the level of soil erosion and/or changing the SDR. With the possible exception of grassed buffer strips, non-structural practices act solely by reducing soil erosion, while structural practices, such as terraces, diversions, sod waterways, etc., generally affect both soil erosion and the SDR.

2.) When comparing the effectiveness of different practices within the same field, the effectiveness of non-structural practices will be directly proportional to their effects on soil erosion. Structural practices such as terraces which trap sediment will usually (but not always) reduce the SDR. Thus their effectiveness for reducing sediment delivery is proportionately greater than their reduction in soil erosion. Structures such as diversion ditches and grassed waterways provide channelized flow from the field to the stream, thus often increase the SDR. In this case, their effectiveness for controlling sediment delivery will be proportionately less than their reductions in soil erosion. In certain cases, these practices may actually increase levels of sediment delivery in fields having an initially low SDR.

3.) The cost-effectiveness of practices on a field, can vary considerably depending on effectiveness, as noted above, and costs which can vary according to type of farm, climate, and physical properties of the field. Thus actual rankings of cost-effectiveness of practices must be determined individually for different types of situations. Four practices were found to have particularly high potential levels of cost-effectiveness. These are conservation tillage, no-till, contouring, and strip-cropping. However, cost-effectiveness for the two reduced tillage systems drops considerably in areas where yield decreases occur with these practices. Strip-cropping is cost-effective only if sod crops are a part of the basic cropping system. If additional sod crops must be grown, the cost-effectiveness of strip-cropping is often quite low.

Conservation tillage in particular, appears to have potential for widespread adoption with little resistance from farmers. Tillage is generally more rapid than with a moldboard plow, and yields can usually be maintained or even increased. Although moldboard plowing is still done on most cropland, the use of conservation tillage appears to be spreading fairly rapidly. Nearly 45% of Iowa cropland now receives primary tillage with either a tandem disc or a chisel plow (James McGrann, Personal Communication, 1978). It should be noted, however, that the use of these implements does not necessarily imply conservation tillage. In some cases several trips over the field are made which buries almost all residue, greatly reducing their effectiveness.

Conservation tillage could be implemented on much if not most cropland in the country with little, if any, cost, other than perhaps a program to convince farmers to reduce their number of field operations.

4). The maximum cost-effectiveness for a terrace system is achieved with one terrace at the lower edge of the field, acting essentially as a sediment basin. If more terraces are constructed to reduce the terrace interval, effectiveness will be increased slightly, but less than the proportional increase in costs. Thus the smaller the terrace interval, the lower the cost-effectiveness of the system.

5). The cost-effectiveness of a farm plan for reducing sediment delivery is highly dependent on the location of practices on the farm as well as the actual practices implemented. If the costs for practices are approximately the same on all fields on a farm, the cost-effectiveness of a non-structural practice on individual fields will be directly proportional to the base level of sediment delivery on the individual fields. Thus a higher cost-effectiveness may be achieved by placing a practice on a field with only a moderate level of soil erosion, but with a high SDR, than on another field with high levels of soil erosion but with a low SDR. The cost-effectiveness of structural practices which change the SDR is proportional to both the initial level of sediment delivery and the initial SDR. Thus if two fields have initially identical levels of sediment delivery, but the first has a high level of soil erosion and a low SDR, while the second has a relatively lower level of soil erosion but a higher SDR, a structural practice will have a higher cost-effectiveness on the second field than on the first.

When a structural practice is implemented on a field, the area above the structure will have a new SDR, while the SDR for the area below the structure remains essentially unchanged. A significant increase in cost-effectiveness for the field can be obtained by placing the most highly erosive crops in the area with the lowest SDR, even though this may have no effect on levels of soil erosion.

6). In general, the higher the degree of effectiveness desired, the lower the resultant cost-effectiveness. This is due to a number of factors including:

   i) Terrace systems are often required to obtain high reductions in sediment delivery, but are not highly cost-effective. Cost-effectiveness of terrace systems declines as terrace intervals become smaller.

   ii) Combinations of practices are generally less cost-effective than individual practices.

The marginal cost of reducing sediment delivery increases slowly at low levels of reduction, but often turns sharply upward at high levels of reduction. The point where marginal cost begins to rapidly increase coincides with the implementation or practices with low levels of cost-effectiveness. Among these practices are intensive terrace systems, substituting sod-crops for row crops, and removing land from prodcution.

7). Optimal farm plans developed for reducing sediment delivery relied heavily on optimal location of practices, and on terrace systems which reduced the SDR. Since these options reduce the sediment delivery proportionately more than soil erosion, it was found that sediment delivery from a farm could usually be reduced by a given amount at less cost than that required to obtain proportional reductions in soil erosion. The primary difference between farm plans for reducing sediment delivery and plans for reducing soil erosion was the choice of fields where practices were implemented and the location of terraces within fields.

8). Current ASCS cost-sharing programs are generally concentrated on structural practices, particularly terraces and permanent sod cover crops. The terrace spacings required by ASCS and SCS are generally considerably less than those found to be optimal in the farm plans. Thus for most cases, little or no cost-sharing money was available for those practices which were most cost-effective for reducing sediment delivery. The net effect of current cost-sharing plans could therefore be that farmers will either not implement any practices or that they will implement practices which have a total cost greater than it would be if other, more efficient practices were adopted. In general, most cost-sharing on non-structural (cultural) practices, especially contouring and reduced tillages systems would be helpful. Also, where technically feasible, cost-sharing on large interval terrace systems would increase the effectiveness of the cost-sharing program.

9). The relative effectiveness of SWCPs for reducing losses of other potential pollutants depends on the classification of the substance as defined in Tables 4-1 and 4-2. In general, farm plans to reduce sediment delivery will reduce losses of these substances less than proportional to the reductions in sediment delivery. Control of solid-phase nutrients and substances classified as strongly adsorbed should be only slightly less than control of sediment delivery; losses of substances classified as moderately adsorbed should be reduced with reductions in sediment delivery, but only slightly. Losses of dissolved nitrogen and non-adsorbed substances will often not be affected by farm plans for reducing sediment delivery, unless these plans include practices, such as sod-based rotations, which reduce required levels of inputs.

10). Although SWCPs and/or incorporation will often be effective in reducing losses of herbicides in runoff, the cost-effectiveness of all practices evaluated was fairly low. All practices reduced edge-of-field losses of herbicides by less than 5 grams per year for each dollar of cost. Thus careful consideration of benefits should be given before implementing these practices solely to reduce herbicide losses.

SECTION 9

THE EFFECTIVENESS OF SWCPs IN COMPARISON WITH OTHER METHODS FOR REDUCING PESTICIDE POLLUTION

Christine A. Shoemaker and Marion O. Harris

The development and successful use of organic pesticides in the 1940's resulted in a new approach to pest control, an approach which seemingly marked the beginning of man's triumph over the traditional enemies of agriculture. The convenience and cost effectiveness of the new chemical methods greatly simplified the task of pest control for the farmer and at the same time conferred on consumers the benefit of high quality, inexpensive agricultural goods. In the intervening thirty years the use of pesticides has continued to increase in both agricultural and nonagricultural areas. At present, about one half billion kilograms of pesticides are used annually in the United States, about half of which are used in agriculture. Use patterns of the major agricultural insecticides and herbicides are listed for various crops in Tables 9-1 and 9-2.

Unfortunately, the use of pesticides in agriculture has created certain environmental problems in both target and non-target areas. Target agricultural areas have suffered the consequences of pest resistence and resurgence and have witnessed the creation of secondary pests and the decimation of beneficial plant and animal species. Agricultural pesticides transported away from target areas have disrupted neighboring as well as distant ecosystems.

Although the movement of pesticides from application sites to surface waters is not entirely understood, the major transport mechanisms are: overland flow, interflow (to ground and surface waters), direct spills or dumping, drift directly into surface waters, and rainout of air-borne pesticides bound on soil particles or in a vapor stage (Leonard *et al*. 1976). Pesticides reaching surface waters may continue to move on through other components of the ecosystem (leaving surface waters through evaporation or food chains) or they may be strongly absorbed or retained until degradation occurs.

SWCPs are only one of many strategies being considered for the control of pesticide pollution of surface waters. Other strategies include improved operational and disposal techniques, changes or improvements in application procedures, and increasing the efficiency of pesticide use through improved pest management techniques. Each of these strategies will obviously have a

TABLE 9-1. USE OF INSECTICIDES ON MAJOR CROPS IN THE U.S., 1971

	1000 kg of Active Ingredients				
Organo-phosphates	**Corn**	**Cotton**	**Soybeans**	**Alfalfa**	**Total Use On Major Crops**[1]
Methyl Parathion	7	10449	1004	62	12528
Parathion	604	1163	27	112	4259
Diazinon	905	-	-	69	1427
Phorate	1210	45	64	10	1899
Disulfoton	142	102	1	103	1840
Malathion	52	305	40	182	1232
Other Organo-phosphates	497	1286	1	186	6373
Total Organo-phosphates	3417	13352	1137	724	29558
Carbamates					
Bux	1625	-	-	-	1639
Carbaryl	750	552	612	47	7542
Carbofuran	1219	-	-	-	1298
Other Carbamates	-	35	-	25	506
Total Carbamates	3594	587	612	72	10985
Organochlorines					
Toxaphene	83	12778	693	8	14940
Other Organo-chlorines	4483	6594	113	229	13185
Total Organo-chlorines	4566	19372	806	237	28125
Other Insecti-cides	839	37	58	20	29049
Total	12416	33348	2613	1053	97717

Source: Adrilenas (1974)

[1] Total use in U.S. on the following crops: corn, cotton, soybeans, alfalfa, peanuts, tobacco, wheat, other grains, other field crops, other hay and pasture, Irish potatoes, other vegetables, citrus, apples, other fruits and nuts, nursery and greenhouse crops.

TABLE 9-2. USE OF HERBICIDES ON MAJOR CROPS IN THE U.S., 1971

Herbicide	1000 kg of Active Ingredients Corn	Cotton	Soybeans	Alfalfa	Total[1]
2,4-D	4156	2	101	105	15115
Propachlor	9682	-	214	16	10786
Alachlor	3800	2	2867	-	6706
Butylate	2645	-	-	-	2689
Atrazine	23636	-	8	-	26007
Simazine	418	-	56	8	783
Arsenicals	-	3440	22	-	3562
Fluometuron	-	1515	-	-	1515
Trifluralin	-	2065	2710	-	5194
Alanap	-	2	1344	-	1515
Vernolate	-	-	657	-	1698
Dinitno Group	7	174	1638	10	3269
Chloramben	20	-	4245	-	4343
Nitralin	-	227	975	-	1230
Other Herbicides	6651	1505	1796	156	83048
Total	51015	8932	16633	295	167460

Source: Adrilenas (1974).

[1] Total use in U.S. on the following crops: corn, cotton, soybeans, alfalfa, wheat, sorghum, rice, other grains, peanuts, sugarbeets, other field crops, pasture and rangeland, Irish potatoes, other vegetables, citrus, apples, all other fruits and nuts, summer fallow, nursery and greenhouse crops.

different effect on the amount of pesticide available for transport away from the target area via drift, runoff, leaching, volatilization, and improper disposal. SWCPs will, in most cases, only affect the fraction of pesticide which is being lost in runoff (solid phase and dissolved) whereas pest control techniques like scouting will bring about an overall reduction in the amount of pesticide being applied, thereby reducing pesticide losses occurring through all transport routes.

The purpose of the study described in Section 9 is to compare the effectiveness of SWCPs with the effectiveness of other strategies in reducing the amount of toxic pesticide reaching the environment. The following items have been critical in this analysis:

1) a delineation of the relationship between pesticide pollution control strategies and the amount of pesticide available for transport via drift, runoff, volatilization, leaching, and disposal;

2) a quantification of the amounts of applied pesticides lost by means of each of these transport mechanisms;

3) a determination of the environmental hazards (toxicity and persistence) associated with pesticide losses occurring through each transport mechanism;

4) a determination of any secondary benefits or hazards associated with the use of each strategy.

Evaluations of the proposed pollution control strategies are hindered by the tremendous ranges in toxicity, persistence, and mobility exhibited by the numerous pesticides presently available to farmers. To reduce this problem to a manageable size, we have examined the effectiveness of the various strategies in reducing the losses of five major pesticides from agricultural lands. The five materials examined are toxaphene, methyl parathion, carbofuran, paraquat, and atrazine. The first three represent each of the major categories of insecticides: organochlorines, organophosphates, and carbamates, respectively. Paraquat and atrazine represent bipyridylium and triazine herbicides. The five pesticides studied are also major weed and insect control agents in corn and cotton, two crops whose production requires a major portion of all pesticides used in U.S. agriculture (Tables 9-1 and 9-2). Fifty percent of the insecticides and eleven percent of the herbicides used in agriculture are used in cotton production. The use of toxaphene and methyl parathion constitutes more than two-thirds of the insecticides used in cotton. The production of corn consumes eighteen percent of the insecticides and fifty percent of the herbicides used in agriculture. Atrazine use makes up for nearly one half of the herbicides used in corn and was identified in 1971 as the most heavily used herbicide in the U.S. (Adrilenas, 1974). Paraquat is another herbicide used in corn production and was studied mainly because of its importance in no till production systems. Carbofuran is a relatively new insecticide used primarily for the control of soil insect pests of corn. Carbofuran and other soil insecticides account for most of the insecticides used in corn production.

In order to relate the conclusions obtained for the five representative pesticides to other pesticides, the following sections include a general discussion of runoff and leaching, air-borne losses, losses due to improper operational and disposal techniques, toxicity, persistence, SWCPs, increasing efficiency in the use of pesticides, and improving operational and disposal techniques. These sections are followed by a more detailed discussion integrating the previous material in an analysis of the representative pesticides. Readers wishing to skim this discussion are advised to read the introductory section entitled, Representative Pesticides, the summary sections at the end of the representative pesticide discussions, and the final conclusions section which includes Table 9-11.

LOSSES OF PESTICIDES FROM TARGET AREAS

Runoff and Leaching

The small percentage of toxic material transported in runoff water and sediment limits the usefulness of SWCPs for reducing pesticide pollution in comparison with other control procedures. Wauchope (1978), in his extensive review of the literature on pesticide runoff losses, concluded that losses are generally 0.5% or less of the amount applied, with larger losses indicating the occurrence of a large runoff-producing rainfall event within one to two weeks after the pesticide application. Wauchope estimated losses for organochlorines to be about one percent of the applied material. He concluded that the major reason for the higher losses is the persistence of organochlorine insecticides in soil. Wauchope also noted that losses of herbicides applied as a wettable powder may be as high as five percent of the applied material, depending on weather conditions and the slope of the treated field. The results of field and modelling studies on runoff are shown in Table 9-3.

The effectiveness of SWCPs in reducing these runoff losses depends in large part upon the chemical and physical characteristics of the pesticide. Since mobility phenomena occur most readily, and in some cases exclusively with dissolved pesticides in the soil water, the balance between solid phase pesticides and dissolved pesticides is the single most important factor in determining the fate of pesticides in soil (Helling *et al.* 1971). Adsorption is often described by the adsorption partition coefficient (Ka) which is defined as the adsorbed concentration on the soil divided by the concentration in the soil water. Under laboratory conditions this value is typically determined by agitating pesticide-free soil with a pesticide in solution, allowing sufficient time for equilibrium, and measuring the concentrations. Under field conditions, the same situation presumably occurs after application as the pesticide goes into solution in the soil water, and eventually reaches equilibrium with the soil. Desorption occurs when an equilibrium soil-water-pesticide solution is diluted with soil, water or both. This would typically happen during and after a precipitation event occurring

TABLE 9-3. RUNOFF STUDIES ON VARIOUS PESTICIDES

Pesticide	Chemical Class	Application Rate (kg AI/ha)	Average Slope (%)	Runoff (% of Applied)
Carbaryl[a]	Carbamate	5.03[1,4,7]	10	0.15
Carbofuran[b]	Carbamate	5.41[1,4,7]	9	0.9
Diazinon[c]	Organophosphate	1.12[1,4,7]	10-15	insignificant
Methyl Parathion[d]	Organophosphate	6.7 [1,6,8]	2	4.8
Toxaphene[e]	Organochlorine	26.8[2,8]	3	0.36
Alachlor[f]	Acetanilide	2.24[1,5,9]	2-9	0.48
Atrazine[g]	s-Triazine	1.2-3.36[1,4,7]	8-22	0.02-5.7
Cyanazine[f]	s-Triazine	2.24[1,5,9]	2-9	0.96
Metribuzin[f]	s-Triazine	0.56[1,5,9]	2-9	0.72
Paraquat[f]	Bipyridylium	1.12[1,5,9]	2-9	3.34
Propachlor[f]	Chloroacetamide	2.24[1,5,9]	2-9	0.21
Propachlor[c]	Chloroacetamide	6.7[1,4,9]	10-15	3.1
Simazine[g]	s-Triazine	1.12-4.48[1,4,7]	8-22	0.04-5.40
Trifluralin[h]	Nitroaniline	1.38[3,6,7]	0.2	0.04

TABLE 9-3. FOOTNOTES

[1] silty loam soil

[2] loamy sand soil

[3] silty clay soil

[4] measurements taken in corn

[5] measurements taken in fields with a corn-soybean rotation

[6] measurements taken in fields with cotton-soybean rotation

[7] application incorporated into soil

[8] foliar application

[9] broadcast spray application

[a] Caro et al. (1974)

[b] Caro et al. (1973)

[c] Ritter et al. (1974)

[d] See Appendix F

[e] Bradley et al. (1972)

[f] Baker et al. (1972)

[g] Triplett et al. (1978)

[h] Willis et al. (1975)

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sometime after application of pesticides. The equilibrium ration of concentrations after the dilution is called the desorption partition coefficient (Kd). A plot of pesticide concentration on soil vs. concentration in water yields the adsorption-desorption isotherm. This isotherm is generally not single-valued (Davidson et al. 1972), that is, $Kd \neq Ka$ for the same soil water concentration under most conditions. Ka and Kd are also dependent on organic carbon content, clay type and content, pH, soil structure, and temperature, as well as the particular pesticide. Thus, adsorption-desorption phenomena are difficult to predict, and are one of the major problem areas in determining pollutant losses from cropland.

Pesticides having strong adsorption characteristics will exist primarily in the solid-phase when located in the soil environment. Paraquat, diquat, and DDT are considered to be in this class of pesticides. Upon contact with soil clay minerals, paraquat and diquat are almost completely and irreversibly adsorbed (Calderbank, 1968). Helling et al. (1971), in their classification of the soil mobility of numerous pesticides, listed DDT as the least mobile. Monitoring studies in the South (Nicholson et al. 1968) have found significant amounts of DDT in sediment, but only insignificant amounts in river water. Toxaphene is similar to DDT in having a low water solubility, and strong adsorption characteristics. Although some studies have indicated that toxaphene is also nearly 100 percent adsorbed (LaFleur, 1974) and highly immobile in soil (Helling et al. 1971), Nicholson has found that toxaphene moves more readily in water than DDT and occurs more frequently in water samples in monitoring studies (personal communication, 1977).

Most other pesticides will exist in both the solid-phase and dissolved states when in the soil environment. Trifluralin, parathion, methyl parathion, phorate, and diazinon are strongly adsorbed (Scott and Phillips, 1972; Saltzman and Yaron, 1971; Meyers et al. 1969), while carbofuran, carbaryl, atrazine, propachlor, and 2,4-D show more moderate adsorption characteristics (Jamet et al. 1974; Leenheer and Ahlrichs, 1971; Meyers et al. 1969; Colbert et al. 1975; Ritter et al. 1974; Newman and Thomas, 1949; Helling et al. 1971). Although no studies have been found on chloramben adsorption, it is apparently one of the most weakly adsorbed pesticides. Helling et al. (1971) classify it as one of the most highly mobile pesticides in soil, while Hilton et al. (1974), state that it "moves readily in sandy soils or following heavy rains."

Pesticide adsorption and desorption is thus a complicated phenomena. A few pesticides are almost 100 percent adsorbed in the soil while others exhibit no significant degree of adsorption. However, the majority of pesticides are in an intermediate class. Because the degree of adsorption for these pesticides depends on the particular pesticide, climatic conditions, soil conditions, and equilibrium time, accurate prediction of the behavior of each individual pesticide in soil may be difficult, and even impossible in certain cases.

Table 9-4 contains several indices which give information concerning the adsorption characteristics of selected pesticides. Some indication of the degree of pesticide adsorption to soil particles is given by a transport mode index taken from Stewart et al. (1975). This index indicates how the pesticide will generally move from the treated field if runoff occurs, that is, whether concentrations of pesticides will be higher in water (dissolved), sediment (solid-phase) or sediment and water equally. It must be remembered that this index only predicts which fraction of the runoff will carry the highest concentrations of pesticides. Since the volume of water lost via runoff is always larger than the volume of sediment lost, a higher percent of the applied pesticide will leave in the dissolved state than in the solid-phase state, in spite of the fact that residue concentrations may be higher in the soil fraction. Paraquat, diquat, toxaphene, and DDT are exceptions to this general rule because of their aforementioned adsorption characteristics.

TABLE 9-4. PESTICIDE - WATER - SOIL - AIR INTERACTIONS*

Pesticide	Transport[a] Mode	Water Solubility[b] (ppm)	Leaching[c] Index	Vaporization[c] Index
Carbaryl	SW	40	2.0	3.0-4.0
Carbofuran	W	700[d]	2.0[e]	0[f]
Diazinon	SW	40[d]	2.0	3.0
Malathion	W	145	2.0-3.0	2.0
Methyl Parathion	SW	60[d]	2.0	4.0
Parathion	S	25	2.0	3.0
Phorate	SW	50		2.0[g]
Toxaphene	S	3	1.0	4.0
Alachlor	SW	242	1.0-2.0	3.0
Atrazine	SW	70	2.0-3.0[h]	2.0[i]
Chloramben	W	700[d]	3.0-4.0[h]	very low[d]
2,4-D	W	900[h]	2.0	1.0
Paraquat	S	very soluble	1.0[j]	nonvolatile[k]
Propachlor	W	580[d]	1.0-2.0[l]	
Trifluralin	S	24	1.0-2.0	2.0

FOOTNOTES FOR TABLES 9-4, 9-5, and 9-6.

[a] Stewart *et al.* (1975) where runoff losses occur from treated fields,

W = chemical moves primarily with water,
S = chemical moves primarily with sediment,
SW = chemical moves with both sediment and water.

[b] Gunther *et al.* (1968) except where noted.

[c] Haque and Freed (1974) estimated leaching and vaporization rates from best available information on loam soil at 25°C under annual rainfall of 150 cm. Data from their table except where noted. Leaching Index number indicates the approximate number of centimeters the material moved through the soil profile.

1 = < 10 cm
2 = < 20 cm
3 = > 35 cm
4 = > 50 cm

Vaporization Index number indicates the kilograms vaporized from one hectare of soil surface during one year.

1 = < 0.1 kg/ha-year
2 = 0.2 to 3.0 kg/ha-year or more
3 = 3.5 to 6.5 kg/ha-year or more
4 = 7 to 14 kg/ha-year or more

[d] Leonard *et al.* (1976).

[e] U.S.E.P.A. (1976).

[f] Harris and Miles (1975) classified insecticides as volatile and non-volatile in soil based on measurements of fumigant activity against insects.

[g] Sanborn *et al.* (1977) give 41% vaporization loss, assume application rate 1.12 kg/ha.

[h] Sanborn *et al.* (1977).

[i] Foy (1963) gives 11% vaporization loss, assume application rate of 2.2 kg/ha.

[j] Fryer *et al.* (1975) except on sandy soils = 2.0 - 3.0.

[k] Calderbank and Slade (1976).

[l] Ritter *et al.* (1974).

FOOTNOTES FOR TABLES 9-4, 9-5, and 9-6 (CONTINUED).

[m] Pimentel (1971) except where noted.

[n] Hilton et al. (1974).

[o] von Rumker et al. (1974).

[p] Eichelburger and Lichtenberg (1971).

[q] Mathur et al. (1976).

[r] F. M. C. Corporation, unpublished data from U.S.E.P.A. (1976).

[s] Lichenstein and Schulz (1964). However, in later work they found that one half of the remaining methyl parathion persisted as bound residues. Work later proved that these bound residues were not excluded from environmental interaction.

[t] Guyer et al. (1971).

[u] Terriere et al. (1966).

[v] Haque and Freed (1974).

[w] Yeo (1967).

Pesticide adsorption characteristics are not the only factors influencing the amount of pesticide leaving agricultural fields in runoff. Other factors are: (1) pesticide properties such as solubility, persistence and volatility; (2) climatic characteristics such as radiation, temperature, and magnitude and timing of runoff-producing precipitation; (3) watershed characteristics such as slope, roughness, management practices, and proximity to water courses; (4) soil characteristics such as texture, pH, antecedent moisture status, hydraulic properties, and organic content; and (5) miscellaneous factors such as timing, mode, placement, formulation, and rate of the pesticide application (Dean and Mulkey, 1979; Stewart et al. 1975; Davidson et al. 1978). Recent work by Bovey et al. (1978) indicates that concentrations of pesticides in runoff water may be reduced if subsurface applications of pesticides are made instead of surface applications.

Because of the difficulties and high costs associated with empirical studies measuring the amount of pesticide found in runoff under varying weather conditions, mathematical models of pesticide movement have become an important tool for estimating pesticide losses. The adsorption coefficient and degradation time of a pesticide are important aspects of these models. This type of mathematical modelling is discussed in earlier sections and in

Appendix F. The results for paraquat, methyl parathion, and atrazine are discussed later in this chapter. Results from several mathematical runoff models are given in Table 9-3.

Very little is known about the extent and environmental significance of the pollution caused by the runoff of pesticides. Incidents of surface water contamination by the organochlorine insecticides are still occurring, and several lakes in the Midwest have been closed to fishing because of accumulations of toxic materials (Luckmann, personal communication, 1978). It is not known whether this contamination is due to the continuing runoff and erosion of soils carrying these persistent pesticides, though many researchers have noted that concentrations of organochlorine residues are higher in areas of high sediment losses and turbidity of surface waters (Glooschenko et al. 1976; Truhlar and Reed, 1976; Schulze et al. 1973; Miles, 1976). A study by Richard et al. (1975) revealed the presence of pesticide residues in the Mississippi River at New Orleans and in every major watershed in the State of Iowa. They found fluctuations in pesticide concentrations in surface water to be consistent with predictions made by the pesticide runoff model of Bailey et al. (1974).

The rate at which pesticides leach through the soil profile is influenced by many of the same factors influencing runoff losses: adsorption/desorption characteristics of the pesticide, chemical reactions, solubility, rate of application, antecedent soil moisture, soil structure, and flow velocity (Sanborn et al. 1977). Such pesticides as 2,4-D, atrazine, cyanazine, dicamba, dimethoate, chloramben, dinoseb, monuron, and methoxychlor all exhibit a greater propensity for movement in the soil. However, Gerakis and Sficas (1974) have concluded that under normal conditions, extensive leaching, and subsequent contamination of groundwater is unlikely.

Leaching will be more of a problem in areas of pesticide disposal, areas with water tables not far beyond the root zone of crops, and in areas with sandy soils containing little organic matter or clay to bind pesticides as they percolate through the soil profile. Some of the reported groundwater contamination by pesticides may occur when solid-phase pesticides are washed down the deep cracks which can appear when heavy rains follow periods of drought (Phillips and Feltner, 1972; Willis and Hamilton, 1973).

Several reports have been made on the presence of pesticides in groundwater. Atrazine and monuron have been found in groundwater during periods of application, and in wet years (Leh, 1968). A study in Iowa revealed pesticide (atrazine, DDE, dieldrin) contamination in all waters originating from shallow wells located in the alluvial plains of contaminated rivers. Current treatment processes for municipal waters did not significantly reduce concentrations of these pesticide residues (Richard et al. 1975).

The most widely documented cases of pesticide contamination of surface waters have concerned the organochlorines. It is not known whether the paucity of pollution reports attributed to runoff and leaching of other pesticides is indicative of field conditions or of the state of the analytical art. In either case, one cannot entirely dismiss the possibility that the runoff and leaching of pesticides from agricultural lands could bring about a low level of pollution in surface and ground waters. This pollution could produce little understood chronic and sublethal toxicity effects which would be hard, if not impossible to monitor (Livingston, 1977).

Air-Borne Losses of Pesticides

In most cases, SWCPs are only effective in controlling the transport of pesticides which have already reached the soil. However, much of the pesticide applied never reaches the soil because it is carried away in the form of drift or because it volatilizes from the leaves of the crop. Pesticides may also volatilize from the soil before they can be washed away by runoff or may be carried on particulate matter eroded by wind. In some cases, the amount lost by drift and volatilization is over 50 percent of the material applied. In such cases, the amount of air-borne pesticide leaving the target area completely dwarfs that being carried in runoff. In other cases, especially for pesticides incorporated into the soil at the time of plowing, the air-borne losses are negligible.

Though it is generally assumed that substantial amounts of pesticides enter the atmosphere by drift, volatilization and wind erosion, little is actually known about the fate of these contaminants once air-borne. There is some indication that pesticides do not remain in the atmosphere for long periods of time, though the rate at which they return to earth depends on degradation patterns during transport, concentration, deposition rate of aerosol particles, and the turbulent exchange coefficient (Duttweiler and Malakov, 1977). Decomposition apparently occurs with air-borne pesticides both in the vapor state and when they are deposited on surfaces (Crosby, 1978). Pesticides not undergoing decomposition may be precipitated in rain or may diffuse slowly into the stratosphere as vapor (Duttweiler and Malakov, 1977).

Work by Cohen and Pinkerton (1966) showed that dieldrin and atrazine residues are associated with suspended solids in rainfall. These data gave the first qualitative evidence that pesticides could enter the atmosphere adsorbed on soil particles, and thus could be transported and redeposited on the earth's surface quite distant from the original area of application. Using reported residue levels of DDT in rainwater samples, Pearce et al. (1978) estimated that about 5000 kilograms of toxic material were deposited by rainfall in the wedge-shaped area covering the Gulf of the St. Lawrence from July to October in 1968. Indeed, Goldberg et al. (1971) have suggested that this aerial fallout may greatly exceed river discharge in its contributions of pesticides to the oceans of the world. The oceans may in turn serve as vast reservoirs, constantly discharging pesticides back into the atmosphere, and thereby serving as a "continual source of replenishment for atmospheric pollution" (Edwards, 1977).

Drift losses of pesticides can be substantial if applicators do not strictly adhere to the operational guidelines intended to prevent most off-target losses. Aerial applications in windy conditions may cause drift damage to surrounding crops, thus one would assume that liable spray applicators would attempt to prevent such contamination. Whether such care is taken to avoid the contamination of surface waters is not known. Factors influencing drift losses have been covered in extensive reviews by von Rumker et al. (1975) and Plimmer (1976). We will only briefly discuss their results here.

Wind velocity, pump pressure, nozzle angle, time of application, nozzle type, droplet size, formulation and nozzle height above the ground all influence the percent of applied pesticide ending up in drift losses (Ware et al. 1969; Ware et al. 1975; Ware et al. 1972; Byass and Lake, 1977). The impact of these factors on pesticide drift losses have been summarized in the extremely informative general drift loss tables presented by von Rumker et al. (1975). Droplet size is extremely important to drift losses. Droplets ranging from 10 to 50 μm in diameter are likely to drift up to several kilometers, whereas droplets greater than 100 μm in diameter rarely drift except in very windy conditions (Gerakis and Sficas, 1974). Brazzel et al. (1968) have suggested that more progress in the control of drift could be made by the precise control of the droplet spectrum than by any other means. Others have pointed out that smaller droplets provide better plant coverage and better insect control (Smith et al. 1975). Obviously, a compromise must be made between application strategies providing optimal pest control and those strategies bringing about minimal drift losses.

The formulation of the pesticide also has a large impact on drift losses. In the past, dust applications of pesticides presented great risks of drift losses and thus, have been largely replaced by spray and granular formulations. In the case of DDT, it was found that dust applications placed fourteen times more material in downwind drift than was found with spray applications (Gerakis and Sficas, 1974). The use of granular formulations greatly reduces drift potential.

The way in which the formulation is applied also influences how much of the pesticide will reach the target area. The potential for drift losses is greatly increased as the distance from nozzle to the target increases. Byass and Lake (1977) have estimated that amounts of pesticides lost in drift can increase by a factor of five when boom heights are increased to levels within the range often used in ground applications of herbicides. Surface incorporation of pesticides allows for the most accurate, though expensive, placement of toxic materials. Aerial applications present the greatest potential for drift problems, resulting in four to five times more drift than occurs from high clearance ground sprayers (Ware et al. 1969). Whatever the application procedure, insecticide applications tend to create greater drift losses than herbicide applications as treatment is at high pressure with small droplet sizes, as compared to lower pressure herbicide applications, resulting in larger droplet sizes. Byass and Lake (1977) have shown, however, that even small drift losses of herbicides can be hazardous. In the case of picloram, drift losses comprising of more than 0.01% of the applied material resulted in significant damage to sensitive plant species located downwind of the spraying operation. Thus, in evaluating the importance of drift losses, one must consider the toxic quality of the drift as well as the quantity produced.

Volatilization can be a major cause of pesticide disappearance from plants and soil. It can contribute to the short term toxicity of many insecticides (methyl parathion), the low efficiency of some herbicides (trifluralin), and environmental contamination by those pesticides not readily degrading in air (Spencer and Cliath, 1977; Quinby et al. 1958). Gerakis and Sficas (1974) have estimated that about one half of the pesticides applied to field crops enter the atmosphere through evaporation from soil and plant surfaces. The rate at which this volatilization occurs depends to a certain extent on the pesticide's vapor pressure, water solubility, concentration, and strength of adsorption. However, environmental factors such as cover crops, soil incorporation, air flow rate, composition of the soil, and penetration into plant surfaces greatly modify the effects of the pesticide's physical characteristics, and give rise to wide ranges of volatility in field situations (Harns and Lichtenstein, 1961; Farmer et al. 1973; Lichtenstein et al. 1970; Spencer and Cliath, 1977). Soil temperatures reaching 50-60°C further encourage volatilization losses of pesticides from agricultural fields (Kearney et al. 1964; Swoboda et al. 1971). The rate of volatilization of pesticides buried in the soil is much slower and depends on the rate of pesticide desorption from soil, the rate of diffusion to the soil surface, and the mass flow of water to the soil surface (Spencer and Cliath, 1973; Farmer et al. 1973). Farmer and Letey (1974) have further discussed the effects of soil incorporation on volatility, and have developed several models for the prediction of volatility losses.

Direct measurement of volatilization under field conditions is quite difficult, thus indications of volatilization losses generally come from either laboratory studies or indirectly from air quality sampling. Lichtenstein et al. (1970) have studied the relative volatilities of various organophosphate and organochlorine pesticides and found that approximately 0.96% of the parathion, 0.63% of the diazinon, 16.3% of the fonofos, and less than 0.01% of the azinphos methyl volatilized from soil incubated at 30°C, whereas 26.1%, 11.5%, 1.32% and 0.78% of aldrin, lindane, dieldrin, and DDT volatilized, respectively, in the same period of time. Haque and Freed (1974) have predicted higher vaporization rates for both parathion and diazinon.

Parathion and methyl parathion have been found in air samples taken in Florida (parathion and methyl parathion) and Alabama and Mississippi (methyl parathion alone), albeit less frequently, and at lower levels than the organochlorines (Stanley et al. 1971). The pervasiveness of organophosphorus residues was further illustrated by the results of another study wherein the air in the homes of people working with pesticides (farmers and formulators) was tested for methyl parathion and parathion. For the pesticide formulators, 25 percent of the indoor samples, and six percent of the outdoor samples contained methyl parathion, while none was found at the farmers' homes. For parathion, 85 percent of the inside sites and 65 percent of the outdoor sites at formulators' homes showed residues, while approximately 50 percent of the indoor sites and 33 percent of the outdoor sites at farmers' homes showed contamination. The researchers commented on these surprising results by noting that the volatility and short environmental half life of parathion would not lead one to expect its occurrence in samples of house dust and inside air (Tessari and Spencer, 1971).

Few studies on carbamate volatilization are available. Harns and Lichtenstein (1961) found no evidence of carbaryl volatilization in tests for toxicity to house flies. However, Haque and Freed (1974) give carbaryl a vaporization index of three to four (see Table 9-4), compared to two for malathion and three for diazinon and parathion. Thus carbaryl losses from soil apparently can occur. Stanley et al. (1971) did not look specifically for any carbamates in atmospheric samples tested, but did check unidentified peaks (gas chromatograph), and mentioned no findings of carbamates.

Volatilization rates of the herbicides propachlor, chloramben, and paraquat are either unknown or considered insignificant, while rates for 2,4-D may vary considerably, depending on the formulation (Hilton et al. 1974). Atrazine losses through volatilization are generally not large, but can be significant if high temperatures and prolonged sunlight follow application prior to precipitation (Hilton et al. 1974; Harris et al. 1968). Trifluralin is subject to both volatilization and photodecomposition, and must be soil incorporated within eight hours after application. Results from Watkinsville, Georgia, show vapor losses of approximately 3.5% of applied trifluralin during application, and a total of 25.9% of the applied amount lost through photodecomposition and volatilization over the growing season (White et al. 1977).

Indices and qualitative data concerning pesticide vaporization are shown in Table 9-4. A vaporization index developed by Haque and Freed (1974) estimates the annual vapor losses of selected pesticides from a loam soil at 25°C, under an annual rainfall of 150 centimeters. According to their index, BHC, heptachlor, parathion, diazinon, and alachlor may have vapor losses of up to 6.5 kilograms per hectare, while carbaryl, methyl parathion and toxaphene may vaporize at a rate of 7 to 14 kilograms per hectare.

Very little is known about the environmental significance of air-borne losses of pesticides from agricultural lands. Although air samples taken over agricultural and nonagricultural areas have indicated the presence of several insecticides, many researchers have dismissed the low residue levels as insignificant. However, the reduced winter hardiness of non-target plants associated with herbicide use, and the mental health problems observed with exposure to organophosphate insecticides may produce environmental hazards which are so insidious as to escape detection. The dangers and presence of herbicide drift have been attested to in a study done by Hibbs (1976), who showed that the decline of the hackberry (Celtis occidentallis L.) in northwest Iowa could be attributed to herbicide drift from corn growing regions. Byass and Lake (1977) have shown that sensitive plant species located downwind of spraying operations can be damaged even when applications are done by ground equipment in the lightest winds encountered. Unfortunately, the transient nature of such drift losses makes it difficult to trace the resultant contamination back through its transport mechanism to the source. Further evaluations of drift and volatilization losses will probably be limited until more data are gathered on the degradation and rate of fallout of air-borne pesticides.

Pesticide Losses Due to Improper Operational and Disposal Techniques

Improper operational and disposal techniques are a serious cause of pesticide pollution of surface waters in agricultural areas. The filling of spray tanks and subsequent disposal of containers often takes place beside ponds or streams, thus, spills from overfilled spray tanks, spills from containers, and the absence of check valves on intake hoses may all result in direct contamination of aquatic systems. In some cases, diluted pesticides remaining in the spray tank after crop treatment are dumped on land or down storm sewers. Disposal problems have also occurred in nonagricultural sectors because of negligence, and the lack of available techniques for disposal of the large quantities of surplus pesticides handled by commerical applicators, chemical manufacturers, and registered pesticide disposal services. The preponderance of disposal-related fish kills in western New York reflects the hazards involved in such procedures and indicates the need for programs and research to help establish safer and more convenient means of preventing such contamination (Walter et al. unpublished).

Several investigations have been done to determine whether pesticides and their containers are being properly disposed of in farming areas. In a study currently being done at Southern Illinois University, 600 five-gallon, used pesticide containers were sampled at one farm disposal site. It was found that less than 40 percent of the sampled containers were rinsed and less than 20 percent were bottom punctured. An average of more than 2.5 ounces of pesticide was found in the nearly 370 cans which were not rinsed. Other used cans of trifluralin, alachlor, 2,4-D, and butylate contained more than 1 quart to 3.5 gallons, 4.25 gallons, 1 gallon and 1 quart, respectively, of undiluted pesticides. Another study at Oregon State University has shown that on the average, nearly 6 ounces of pesticide is left unrinsed in five-gallon containers (Luckmann et al. 1978).

ENVIRONMENTAL IMPACT OF PESTICIDE LOSSES

Toxicity

It is important to consider not only the amounts of material leaving target areas but also their impact on non-target organisms. Most researchers have attempted to evaluate these environmental effects through acute toxicity bioassays. However, while these laboratory tests have been valuable for a comparison of the relative toxicities of various pesticides, their results are often of limited significance without verification from field tests (Livingston, 1977). These lab experiments, by their very nature, leave out many complex environmental factors which can influence the uptake and potential effects of pesticides. In the case of aquatic systems, pesticide adsorption to sediments and vegetation could greatly reduce toxicity, while sublethal and secondary effects, concerning destruction of an important food species, could end up being as important as the direct toxic action itself (Holden, 1973; Hurlbert, 1975). Potential destruction of food species is particularly important in studies of herbicides, for they, by the very nature of their action, are likely to affect the lower trophic levels of food chains (von Rumker _et al_ 1975), and could conceivably have complex secondary effects related to the presence of decaying vegetation and altered water quality (Livingston, 1977). Another factor, which acts to further complicate any evaluation of environmental impact, is the possible toxic synergism of pesticides with other pesticides and pollutants present in the aquatic environment (Lichtenstein _et al._ 1969; Lowe _et al._ 1970).

Livingston (1977), after having considered the limitations of laboratory data, suggested that "if there is one major need in pesticide research today, it is the completion of the natural sequence of hypotheses based on experimental work through field testing and ecological analysis." Hurlbert (1975) in turn, has criticized present attempts at experimental field investigations and suggests that if their conclusions are to be taken seriously in the future, proper consideration must be given to adequate controls and replications. Whatever the approach, it is evident that more research must be done to ascertain the effects of pesticide use on both terrestial and aquatic systems, for as von Rumker _et al._ (1975) put it, "it is not known whether or not currently practiced monitoring and observation methods would detect such effects prior to the occurrence of massive ecological damage."

Having stated the limitations of such data, we present lethal dose and lethal concentration ranges in Table 9-5. A number of insecticides are classified as highly toxic. These include aldicarb, parathion, methyl parathion, carbofuran, phorate, diazinon, and toxaphene, the latter three being extremely toxic to fish. At a toxicity of 0.65 mg/kg for aldicarb, less than 0.2 g would be fatal to an average size (70kg) human. Thus special care is required for handling these materials, particularly since most are toxic by oral, inhalation, and dermal routes. Herbicides, with the exception of paraquat, are relatively nontoxic to mammals. Effects on fish, with the exception of trifluralin, are also minor when compared to insecticidal toxicity, however as mentioned previously, sublethal and secondary effects may adversely influence fish populations. Other materials such as simazine, diuron and linuron are extremely toxic to algae (Sanborn _et al._ 1977).

Persistent chemicals are sometimes stored in the cells of organisms and are passed on to predators in higher trophic levels in a food chain. Organochlorine insecticides such as DDT have been found in much higher concentrations in the bodies of members of higher trophic levels than in the bodies of the lower organisms which originally ingested the material (Metcalf _et al._ 1971). This phenomenon is known as biomagnification.

Few studies have been done on biomagnification of organophosphates and carbamates. Miller _et al._ (1966) showed short term concentration of parathion and diazinon in mummichog and mussels. Concentrations in the animals were as

TABLE 9-5. TOXICITY OF VARIOUS PESTICIDES[m]

Pesticide	Mammals LD$_{50}$ mg/kg	Fish LC$_{50}$ ppm	Birds LD$_{50}$ mg/kg
Carbaryl	200-540	1.5-2.0	1000-3000
Carbofuran	8-14[b]	0.21[b]	0.24-5.0
Diazinon	108	0.052-1.45	3.5-4.3
Malathion	480-1500	0.02-12.9	1485-5000
Methyl Parathion	6-32	2.75-9.0	7.5-10.0
Parathion	4-56	0.125-2.7	0.12-24
Phorate	3.7	0.005-1.0	6.2-7.1
Toxaphene	15-240	0.002-0.05	10-316
Alachlor	1800[b]	2.3[b]	
Atrazine	1750-3080	0.55-12.6	> 2000
Chloramben	3500-5620[b]	7.0[b]	> 1000
2,4-D	100-1000	1-1000	970-4048[h]
Paraquat	150	45-840	
Propachlor	710[n]	735-5000[n]	
Trifluralin	> 2000	0.001-0.540	> 2000

Footnotes defined on pages 215 and 216.

high as 24 times the water concentration, but dropped considerably within six days. Some organophosphates, such as parathion have been found in small concentrations in sea animals (Deubert and Gray, 1976). However, because of their low fat solubility, rapid degradation in biological organisms, and other chemical and physical properties, it is unlikely that organophosphorus insecticides biomagnify in food chains (USEPA, 1975). Sangha (1972), reporting on carbamates in a model ecosystem, found propoxur and carbofuran to be degradable, but had persistent residual products, whereas mobam and aldicarb exhibited low biodegradability. Mobam was concentrated from 9 ppb in water to 15400 ppb (parts per billion) in mosquito larvae, and 110 ppb in fish, while aldicarb had a concentration of 7.4 ppb in water, 17000 ppb in mosquito larvae, and 2320 ppb in fish.

Biomagnification has generally not been a problem with most herbicides (Livingston, 1977). Metcalf and Sanborn (1975) have tested alachlor, atrazine, dicamba, cyanazine, 2,4-D, propachlor, and metribuzin in a model ecosystem, and have concluded that their use will not result in biomagnification in the aquatic food chain. They also tested Counter, which showed a low level of ecological magnification in aquatic systems. Mauck _et al_.(1976) have found that simazine residues can persist for over a year in aquatic fauna, with biomagnification occurring in invertebrates. However, they found no significant harmful effects over a two-year period of studying pond fauna. An exception to the general rule of herbicide nonaccumulation is trifluralin, which has been shown to magnify in aquatic food chains (Sanborn _et al_. 1977).

Problems have arisen with herbicides and the organophosphate and carbamate insecticides in spite of their general low levels of biomagnification. Adverse effects have occurred at levels below the established acutely "safe" level. Leptophos has been shown to produce delayed neutrotoxic effects on chickens and was responsible for the deaths of 1500 water buffalo in Egypt after inadvertent exposure. Recent evidence that DBCP, a pesticide ingredient previously thought to be safe, can cause sterility and possibly cancer in humans at long-term, low-level exposures has brought into question the adequacy of the whole procedure for testing pesticide safety (Stevens, 1977). Further research on neurological and other types of low-exposure chronic effects, particularly from organophosphorus insecticides, has also been called for (Metcalf and McKelvey, 1976).

Persistence

The term persistence, as applied to pesticides, has come to have several meanings perhaps implicit in the literature, but nonetheless confusing for the layman. One meaning commonly used by researchers defines persistence by the presence or absence of insecticidal or herbicidal activity. Highly adsorbed pesticides losing toxicity on adsorption to soil particles would, by this definition, be described as nonpersistent. A second definition might lead other researchers to studies of the period of time necessary for a complete disappearance of the original material, be its toxicity continued or discontinued. Highly adsorbed pesticides would be persistent according to this definition. Concern about toxic degradation products has led to the third, and what is perhaps the most inclusive of all persistence definitions, that is, persistence as that period of time necessary for a complete degradation of the pesticide and its metabolites into harmless products. Researchers using this definition have often expanded what was simply a determination of soil and water persistence, to a consideration of persistence in the total environment, realizing that pesticides disappearing from soil through volatilization may go on to cause pollution problems through subsequent atmospheric cycling. Unfortunately, research using this last definition has been limited by inadequate analytical techniques, and by a lack of data concerning the persistence of pesticides and their metabolites in soil, water and air.

Until recently, pesticides were generally classified as persistent or nonpersistent. The chlorinated hydrocarbons, which are generally considered to be persistent, have degradation rates often measured in years, while the "nonpersistent" pesticides generally have degradation rates measured in weeks or months. Recent evidence of the existence of "bound residues" of pesticides existing in soil, makes these classifications considerably less distinct. This appears to be particularly true for the organophosphorus insecticides, previously considered non-persistent (Davidson, personal communication, 1977). Lichtenstein et al. (1977) tested an agricultural loam soil for methyl parathion and fonofos residues and found that after 28 days, seven percent of applied methyl parathion and 47 percent of applied fonofos existed as extractable residues. However, an additional 43 percent of the methyl parathion and 35 percent of the fonofos existed as bound residues (measurement was by radio-carbon techniques, thus percentages include both the pesticide and possible metabolites). Similar results have been reported for parathion (Katan et al. 1976). Lichtenstein concludes: "In view of the above findings, the expressions 'disappearance' and 'persistence' of pesticides so widely used during the last two decades should be reassessed to consider the bound products." The possible existence of bound residues should be kept in mind when interpreting the results from the literature quoted below.

Persistence data for various selected herbicides and insecticides is given in Table 9-6. When looking at this data, it must be remembered that each figure represents one set of interactions between the pesticide, agronomic practices and biological conditions. A more detailed review of persistence can be found in the work done by Sanborn et al. (1977).

The persistence of the organochlorine insecticides is attested to by the pervasive nature of their residues in agricultural soils throughout the country. Some of the crops grown on these soils continue to be contaminated by pesticide residues and metabolites in spite of the general decline in organochlorine use in recent years (Moore et al. 1977). The contamination resulting from these persistent residues is not isolated in soils and crops; wind and water erosion of agricultural fields provide ready transport, introducing toxic materials to aquatic and terrestial ecosystems far from the area of original application. Moore et al. (1977) have found dieldrin residues in soybeans grown on fields previously treated with aldrin and in soybeans from untreated fields. They suggest that the dieldrin contamination in untreated fields is due to the transport of contaminated soil particles by air

TABLE 9-6. PERSISTENCE OF VARIOUS PESTICIDES

Pesticide	In Soil	In Water
Carbaryl	3 weeks[o]	1-2 weeks in river water[p]
Carbofuran	16 weeks[q]	1 week[r]
Diazinon	8-16 weeks[o]	
Malathion	< 2 weeks[o]	2-4 weeks in river water[p]
Methyl Parathion	4 weeks[s]	2-4 weeks in river water[p]
Parathion	1-834 weeks[h]	4-8 weeks in river water[p]
Phorate	1-104 weeks[h]	
Toxaphene	5 years[t]	1-5 years[u]
Alachlor	8-12 weeks[o]	
Atrazine	16-48 weeks[v]	
Chloramben	4-8 weeks[h]	half-life 6 hours[h]
2,4-D	1-4 weeks[o]	9 days[h]
Paraquat	48 weeks or longer[m]	1/2 hour[w]
Propachlor	3-4 weeks[w]	
Trifluralin	12-16 weeks[o]	

Footnotes defined on pages 215 and 216.

or water. Menzie (1972) lists the half-lives of several organochlorine insecticides in soils as three to ten years for DDT, one to seven years for dieldrin, seven to twelve years for heptachlor, two to four years for chlordane, and ten years for toxaphene.

Much research has been done on the persistence of organophosphate insecticides. Lichtenstein and Schulz (1964) investigated parathion, methyl parathion, malathion, and the following parathion metabolites: para-oxon (the main oxidation product), aminoparathion, p-nitrophenol, and p-aminophenol. They found that five percent of the parathion remained two and one half months after application, whereas, malathion degraded to the same percent of the applied material in four days. Para-oxon degradation in soil was relatively rapid with 50 percent of the material being lost within 5-1/2 hours, and 90 percent gone 24 hours after it was applied. Another experiment showed p-nitrophenol to be a major degradation product of para-oxon; as para-oxon residues decreased over a 24-hour period, p-nitrophenol residues increased proportionately. The p-nitrophenol disappeared completely within sixteen days, amino-parathion within three days, and p-aminophenol almost immediately. Stewart et al. (1971) have found residues persisting for up to sixteen years when parathion was applied at high application rates (35 kg/ha). Diazinon disappears relatively rapidly in most soils. Getzin and Rosefield (1968) found that in a moist Sultan silt loam, 45 percent of the applied diazinon had degraded in four weeks. Diazinon can be more persistent, though problems are often limited to those situations when the material is sprayed on a cold, dry, alkaline soil which has never been treated with the insecticide before (Sanborn et al. 1977). Experiments carried out on phorate on the same soil showed that only 20 percent of the initial concentration remained after four weeks (Getzin and Shanks, 1970). However, a review by Sanborn et al. (1977) reported studies wherein phorate's persistence in soil extended up to 104 weeks.

Herbicides range from the relatively persistent to those disappearing within weeks. Horowitz et al. (1974) rated trifluralin as moderately persistent, compared with ten triazine, diazine, and substituted urea herbicides. At a high concentration rate of application (7.2 ppm), trifluralin continued to cause oat injury up to five months in the glasshouse. In the field, trifluralin remained phytotoxic for six months, except at the highest rate of application (14.7 kg/ha), which showed residual activity after nine months. The use of simazine on fruit and vegetable crops has created some soil residue problems for susceptible crops following treated crops, as has the use of atrazine on field and sweet corn (Sweet, personal communication, 1978). The persistence of alachlor is similarly marked in months, though its use presents no problem to sensitive crops grown in the following season (von Rumker et al. 1974).

Propachlor, 2,4-D, and chloramben all seem to have a somewhat lower persistence. Most of the 2,4-D applied to soil disappeared within a month (Audus, 1952; Newman and Thomas, 1949), while approximately 30 percent of applied propachlor remained in the soil 21 days after application (Ritter et al. 1974). Although no studies on chloramben degradation were found, Hilton et al. (1974) report average persistence of six to eight weeks in the field. Paraquat's complex soil adsorption behavior creates a situation wherein it is difficult to label it as either persistent or nonpersistent. This phenomenon will be further discussed in the section on paraquat.

As can be seen from Table 9-6, persistence data for pesticides in water are scanty. This information is extremely important to any evaluation of the importance of nonpoint sources of pesticide pollution from agriculture. The behavior of the toxic material in aquatic systems can be such that it may be imperative or entirely pointless to try to prevent small concentrations from entering surface waters.

Numerous cases of contamination of aquatic environments by organochlorines have been documented, an occurrence which might be expected given the persistence of these materials. Eichelburger and Lichtenberg (1971), in their persistence studies using river water, found that 100 percent of the heptachlor epoxide, endrin, dieldrin, DDT and DDE remained after eight weeks. Wolfe et al. (1977) studied the hydrolytic degradation of methoxychlor at pH's and temperatures commonly found in aquatic systems, and found its half-life to be approximately one year. Organochlorines generally degrade slowly in aerobic aquatic environments, but are aided in their decomposition by microorganisms in anaerobic aquatic environments (Luckman et al. 1978).

The few data available on other pesticides indicate a more rapid degradation in water than is found with the organochlorines. The rapid hydrolysis of organophosphate insecticides in aquatic systems (eight to twelve days) greatly reduces any hazards associated with their use (Pionke and Chesters, 1973; Muirhead-Thomson, 1971). Parathion does not hydrolize as rapidly as the other organophosphates, but undergoes microbial degradation in both anaerobic and aerobic aquatic environments (Graetz et al. 1970). Studies by Harris and Miles (1975) have found residues of parathion, diazinon, and ethion in waters draining Bradford Marsh, an agricultural area which received as many as 19 applications of parathion in one season for the control of onion maggot. The degradation of the carbamate insecticides depends on the pH of the solution, with degradation occurring in less than four weeks in slightly alkaline river water (Eichelburger and Lichtenberg, 1971). Yu and associates (1974) have reported that carbofuran hydrolizes rapidly in water. The degradation of herbicides is influenced by temperature and the availability of sediments containing active microbial populations (Luckman et al. 1978).

STRATEGIES FOR REDUCING POLLUTANT LOSSES

The Effectiveness of SWCPs in Reducing Pesticide Transport

The effectiveness of SWCPs in reducing the runoff losses of pesticides which are moderately adsorbed to soil particles has been addressed in other sections of this report. As our intent in Section 9 is to compare the effectiveness of SWCPs with other methods for reducing losses of pesticides, we will only summarize the findings of Sections 4 and 7 which are pertinent to our discussion.

a. Most pesticides used in agriculture are moderately adsorbed to soil particles and are transported primarily in the dissolved phase rather than the solid phase of runoff. The pesticide residues which are transported in the solid phase are primarily associated with fine clay particles. As a result, SWCPs are not as effective in reducing the movement of these pesticides as they are in reducing the movement of sediment.

b. The persistence of a pesticide has a major impact on runoff losses. In Section 7 it is reported that substitution with a less persistent pesticide will reduce runoff losses more than any of the SWCPs examined.

c. In most cases, SWCPs do not increase the fraction of applied pesticide which is lost in surface runoff.

Before one can conclude that SWCPs are beneficial for the control of the pollution caused by the runoff of moderately adsorbed pesticides, several factors must be considered. The first is that some SWCPs such as no-tillage are associated with an increase in pesticide usage (Section 8). Because of these increased application rates, the amount of pesticide transported in runoff from no-tillage fields can be larger than the amounts transported in runoff from conventional tillage. Secondly, we must consider the possibility of increased leaching of pesticides with SWCPs. Although laboratory, field and modelling studies (Section 7) have generally dismissed the possibility of significant leaching losses, monitoring studies have found pesticides in groundwater and shallow wells (LaFleur et al. 1973; Leh, 1968; Richard et al. 1975). The increased percolation of water brought about by SWCPs could make this pollution problem more serious.

The use of SWCPs for reducing non-point source pollution of surface waters by pesticides is more effective in the case of the strongly adsorbed, persistent organochlorine insecticides and bipyridylium herbicides. These materials are transported mainly in the sediment fraction of runoff, and degrade slowly in surface waters. It has been surmised that SWCPs would keep these toxic materials in agricultural soils where they would degrade without harming species living outside of the target area. However, several factors make these arguments for the implementation of SWCPs to control runoff losses of strongly adsorbed pesticides less compelling. First of all, most of the organochlorines have been taken off the market, leaving farmers with moderately adsorbed pesticides which are transported primarily in the dissolved phase rather than the solid phase of runoff. The use of the SWCP strategy for the strongly adsorbed pesticides is thus limited to pesticides remaining on the market such as toxaphene and paraquat, and the residues of the formerly used organochlorine insecticides remaining in the soil environment. Case studies on paraquat and toxaphene which follow this discussion describe the effectiveness of SWCPs for reducing losses of these materials to non-target ecosystems. The question of organochlorine residues remains and poses some difficult problems. Is it worthwhile to implement SWCPs for the purpose of reducing runoff losses of pesticide materials whose residues remain in the soil but whose use has been much restricted since the early 1970's?

Several facts throw some light on this question. As mentioned previously in Section 9, runoff is not the solitary or primary transport route for organochlorine residues. Many of these residues leave agricultural fields through volatilization, drift, and wind erosion of contaminated soil particles. These air-borne losses may be degraded in the atmosphere or redeposited on the earth. Pearce et al. (1978) have estimated that about 5000 kilograms of DDT were deposited by rainfall during a four-month period in the wedge-shaped area covering the Gulf of St. Lawrence. Moore et al. (1977) have found dieldrin residues in untreated fields, and have related the contamination to volatilization, drift, and wind erosion losses of dieldrin as well as runoff from treated fields. A second point to be made about the soil residues of banned organochlorines concerns their degradation in soil. Lichtenstein et al. (1971) investigated the soil residues of aldrin, heptachlor and their metabolites or formulation impurities in cultivated and non-cultivated fields, and found that the cultivated fields had 76 to 82 percent fewer residues. Runoff losses apparently did not occur (Lichtenstein, personal communication, 1979). Thus the authors attributed the reduced amounts of residues to the breakdown and volatilization of toxic materials. If this prolonged persistence in non-cultivated fields does indeed occur, the reductions of runoff losses gained by such SWCPs as no-till and conservation tillage might be negated by the prolonged persistence of organochlorine residues. As discussed in Section 7, reduced persistence can decrease runoff losses more than SWCPs. A final point to be made about these organochlorine residues is that they have been present in soils for several years now and have to a certain degree already been carried to streams and lakes by runoff or entered the atmosphere by volatilization. Levels of organochlorine residues in wildlife continue to decline (Butler and Schutzmann, 1978), thus it is questionable whether it would be worthwhile to implement SWCPs for strongly adsorbed pesticides at this late date.

The remainder of the material in Section 9 focuses on the comparison of the effectiveness of SWCPs with the effectiveness of other practices for reducing pesticide transport away from target areas. In many cases, it will be shown that there are alternatives which are more effective than SWCPs in reducing pesticide pollution. However, it must be noted that many SWCPs implemented for nutrient and sediment control will have additional benefits in reducing pesticide losses. In some cases SWCPs will serve a dual role as pollution control strategies and as pest control strategies. Sod-based rotations and strip-cropping will, in certain situations, reduce the amounts of pesticide being applied to the crop and furthermore reduce any pesticide runoff losses that might occur. Shelter belts serve a similar dual function by reducing drift and wind erosion losses of pesticides from agricultural lands. SWCPs which do not increase pesticide use can also be used in combination with practices like scouting which reduce the number of pesticide applications needed for crop production.

Unfortunately, little is known about the impacts of various SWCPs on weed, insect and disease populations. For example, SWCPs such as no-till and conservation tillage markedly alter the crop ecosystem and can, thus, have a significant impact on pest population dynamics. As was pointed out earlier, these alterations require increased usage of herbicides and allow for a build-up of crop residues in fields. There is some concern that insect and disease problems will be exacerbated by the continuous presence of this protective crop cover. Similarly, other SWCPs could potentially have an impact on the survival of insect and pathogen pests because of the increases in soil moisture accompanying reductions in runoff losses. Until further research is done in these areas, we cannot be sure that the implementation of SWCPs will not increase pest problems in some cases and thus also increase the amounts of pesticide applied and transported from fields.

Increasing Efficiency in the Use of Pesticides

Because of the pest resistance and hazards to non-target organisms associated with pesticide use, extensive efforts have been made to develop effective pest control programs which require reduced amounts of pesticides. These programs increase efficiency of pesticide use by implementing one or more of the following strategies:

(1) pest monitoring to improve the timing of pesticide applications relative to the pest population,

(2) biological or cultural pest control methods to supplement or replace the control given by pesticides,

(3) changing application procedures or equipment to improve the placement of pesticides.

Strategies (1) and (2) have been developed and implemented primarily for the management of insect and disease populations, whereas, strategy (3) deals with placement problems which are common to the application of insecticides, fungicides, and herbicides. As there has been little integration in the research and development of the first two strategies with the third strategy, we will discuss their potential contributions to pollution control separately.

Population monitoring indicates when disease and insect populations reach densities which warrant a pesticide treatment. This density is called an "economic threshold" or "the density at which control measures should be applied to prevent an increasing pest population from reaching the economic injury level" (Stern et al. 1959). The economic injury level is the pest density, p, such that,

$$C = Y(p) - Y'(p)$$

where C = cost of the pesticide application
Y = net economic return from the harvested crop given that the

pest density is p and a pesticide application is made

$Y'(p)$ = net economic return from the harvested crop given that the pest density is p and a pesticide application is not made.

These carefully timed treatments can be contrasted with the formerly used, fixed-schedule spray treatments which occurred at regular intervals throughout the growing season, whether or not the pest population was present at injurious levels. Pest population monitoring has in most cases reduced the number of pesticide treatments applied in comparison with these fixed-schedule spray programs. A national survey of entomologists indicated that if the population monitoring technique of scouting was implemented in all areas where it was cost effective (*i.e.*, where scouting causes no net reduction in farm income), the national use of insecticides in cotton would be reduced by approximately 20 percent (Pimentel *et al*. in press).

Cultural methods of pest management alter the environment of the pest in order to reduce the potential for pest damage and the need for pesticide. Examples of cultural methods of control are changes in planting and harvest dates and the destruction of crop residues serving as overwintering sites for insect pests. Pests which cannot complete their life cycles in the absence of a continuous crop of their host plant are often successfully controlled by crop rotations.

Biological control methods take advantage of the pest control provided by the various natural enemies of insect and weed pest populations. In some cases, predators or pathogens are released repeatedly throughout the growing season to provide short term control similar to the control given by pesticides. The most widely used of these short-lived biological agents is *Bacillus thuringiensis* which is used for the control of a range of lepidopterous insects. Much effort has also been directed at establishing more permanent biological controls by encouraging the growth of predator or parasite species which can maintain population levels large enough to contain pest populations for many generations. One of the most successful examples of biological control is the suppression of olive parlatoria scale by two hymenopterous parasites. Prior to the introduction and establishment of these parasites, olive parlatoria scale was the major insect pest of olives. It is estimated that these parasite introductions have saved olive growers over seven million dollars in insecticide costs (Huffaker *et al*. 1976). In addition to the introduction of exotic parasite species, attempts have been made to protect native populations of beneficial insects by changes in pesticide formulation, dosage or application timing. Beneficial insects can also be protected by habitat management practices such as the strip-cropping of alfalfa.

Resistant plant varieties represent another important means of reducing pesticide use in agriculture. Wheat varieties resistant to Hessian fly and corn varieties resistant to European corn-borer have been available for years and have been tremendously important in reducing the need for insecticides in wheat and corn production. Cotton varieties such as frego bract cotton with resistance to boll weevil, are currently being developed for commercial release. The use of resistant plant varieties is more widespread for disease control than it is for the control of insect damage.

Recent efforts in insect pest control have concentrated on the integration of biological controls, cultural controls, resistant plant varieties, and pesticide treatments timed by population monitoring. Approaches which attempt to coordinate several methods of control are referred to as "integrated pest management programs" or "IPM". The most effective combinations of control methods vary for different crop species and localities. For example, in Texas cotton pests are controlled by an IPM program which consists of a combination of scouting, short season cotton, and a "diapause control" program for destroying overwintering sites. "Diapause control" requires the destruction of crop remains in crop fields and adjacent vacant fields followed by insecticide treatments to kill overwintering boll weevils. Such a program involving scouting and sanitation has been implemented on thousands of hectares of cotton in the lower Rio Grande Valley region. Frisbie has estimated that the implementation of this program has allowed farmers to reduce insecticide applications from twelve treatments to nine treatments per year (reported for 1972-1974 in Pimentel et al. in press). Because of differences in climate and pest complexes, IPM programs for cotton in the West and in the Southeast are quite different from the programs used in Texas, but in all areas there are IPM programs which require reduced amounts of insecticide at no loss in income to the grower.

To determine the national impact of the implementation of integrated pest management techniques in cotton production, Pimentel et al. (in press) interviewed 32 insect pest specialists working in cotton producing regions. The results of this survey indicate that the implementation of IPM techniques in cotton would reduce insecticide use by an average of 40 percent without decreasing farm income (Table 9-7). Lacewell et al. (1976) studied an IPM program in Texas which utilized narrow row, short season cotton and found that the program required 27 percent fewer insecticides and herbicides. In addition, they found that production costs, in terms of dollars spent per kilogram of lint produced, were reduced from $1.04 for typical Frio County production systems to $0.593 using the IPM program. Energy use on a per hectare basis was reduced 33 percent, while energy use per kilogram of lint produced was reduced by 56 percent.

Integrated pest management programs are also being developed for insect pests of corn. The major focus of this research has been on the control of the insects which cause the most damage to corn, that is, the corn rootworm and other soil inhabiting insects. Scouting has a great potential for reducing the amounts of insecticide needed to control these soil inhabiting pests. Luckman reported that at least 50 percent of the fields scouted in his field studies in Illinois had corn rootworm population densities below the economic threshold, and thus were not treated with a soil insecticide (personal communication, 1978). Before scouting programs were implemented, fields were routinely treated every year to control the corn rootworm. Since the cost of scouting is less than half the cost of an insecticide treatment,

TABLE 9-7. EFFECT OF NATIONWIDE IMPLEMENTATION OF COTTON INTEGRATED CONTROL PROGRAMS ON INSECT CONTROL COSTS AND ON INSECTICIDE USE (From Pimentel *et al.* in press)

	Insect Control Alternative	Savings Insect Control Costs (Million $)	Insecticide Use as a Percentage of Current Use[1]			
			Chlorinated Hydrocarbons	Organo-phosphates	Carbamates	Total
1.	Scouting Only	26	82	72	45	77
2.	Most Economical IPM Currently Available	81	61	54	98	59
3.	Most Economical IPM Available in 5-10 years	136	41	40	88	42

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[1] Using insect control practices and cotton acreages reported in 1972-1974 it was estimated that insect control costs were $253 million per year and the use of chlorinated hydrocarbons, organophosphates and carbamates in cotton was 46, 31, 1.3 million kilograms, respectively.

scouting is a cost-effective means of reducing insecticide use in these areas.

Cultural control methods have also proven to be an effective way of reducing the need for chemical control of the corn rootworm. Rotation of corn crops with other crops disrupts the corn rootworm's life cycle, and prevents any rootworm damage from occurring in the corn crop following the rotation crop. When rotations involving more than one year of corn are used, rootworm damage may occur during the second or subsequent years following a rotation. Sod-based rotations can thus be seen as both a SWCP and a pest management technique. Unfortunately, in some areas the use of crop rotation is quite costly for the farmer because the net income from the alternate crop (e.g., alfalfa or soybeans) is considerably lower than the net income for corn.

In summary, recent advances in the integration of pest control methods have resulted in cost-effective programs which reduce the need for insecticides in cotton, corn, and other crops. Wider implementation of these programs promises substantial reductions in insecticide use and corresponding reductions in air-borne, runoff, and leaching losses of insecticides. The development of integrated pest management programs has been slower in the case of weeds, although expanded research efforts in this area are currently underway. It is impossible to predict the impact of this IPM research on the future of herbicide use in U.S. agriculture.

Given the large losses of pesticides to non-target areas, it is indeed regrettable that more has not been accomplished to improve the placement of pesticides relative to the pest species. von Rumker et al. (1974) estimated that less than one percent of the insecticide applied by aerial equipment is absorbed by insects through contact, inhalation, and ingestion. The relatively low price of pesticide has reduced the incentive for research and development in this area. Himel (1974) described present systems of crop spraying as the most inefficient industrial process ever practiced.

In spite of the lack of research on new spraying technologies, there are several simple techniques which are presently available to improve the placement of pesticides. As discussed in the section Air-borne Losses, proper care and adjustment of spray equipment could vastly improve this situation. Pumps, agitation systems, and pressure regulators should all be in proper working condition, and nozzle tips should be regularly checked for wear. Tests have shown that wettable powder formulations quickly wear down nozzle tips creating situations wherein application delivery rates increase by twelve percent after spraying just 50 acres (Fruit Grower, 1978). Chemical rate, tractor speed, nozzle size, line pressure, height of the boom and band width all influence the placement of the pesticide and thus will greatly affect the effectiveness of the material. Proper timing of spray procedures is also extremely important to any improvement in the placement of pesticides. Spraying in the early morning or at night will substantially reduce drift losses to non-target areas (Smith et al. 1974; Ware et al. 1972). The use of adjuvants to increase the viscosity of spray solutions will further reduce drift losses. Richardson (1974) tested two adjuvants and obtained 63 to 98 percent reductions in drift losses.

While the development of new spray equipment has been slow in coming, several researchers have reported on promising new technologies. Ware et al. (1975) have shown that under identical weather and application conditions, aircraft using Raindrop™ nozzles deposited 25 percent more spray on target, and reduced off-target drift by 50 percent when compared with type D flooding nozzles. McWhorter (1977) has reported on the use of the recirculating sprayer for post-emergence control of johnsongrass, redroot pigweed, and hemp sesbania in soybeans, and has found a spray recovery of 60 to 90 percent, which is then recirculated for use. Work at the University of Minnesota has been done to evaluate the roller applicator which has a frame-mounted, carpet-covered roller replacing the header of a self propelled swather. This roller improves the placement of pesticides because the active material is fed onto a roller which does not drip pesticide on the crop. The unit has been tested using Roundup to treat quackgrass and reed canary grass in blue grass seed fields, however, the principle may have an application in many crops (Sein, 1978).

Improving Operational and Disposal Techniques

Extension personnel have for many years made available information pertaining to the proper techniques for filling spray tanks and disposing of concentrated and dilute pesticides and their containers. This information has been received with varying degrees of interest on the part of farmers. Pesticide experts working in the field have noted disposal and filling techniques ranging from very poor to excellent and have called for an expansion of education programs to further acquaint farmers with proper techniques and hazards resulting from improper disposal. Deposits on pesticide containers might give farmers further incentive to properly rinse and return containers to dealers for recycling. If compliance with voluntary programs is not adequate, expanded regulations concerning disposal should be considered.

While much of the responsibility for proper disposal and operational techniques lies with the farmer, large improvements in the situation are somewhat dependent on the development of new technologies. For example, although some surplus pesticides and empty containers may be returned to the manufacturer for reuse, others will have to be disposed of in a manner which protects the environment and complies with various Federal and State laws. Unfortunately, the technology for such disposal is not highly developed. Incineration is perhaps the most promising method for disposal (Sanborn et al. 1977) but is presently limited in its applicability by high costs and the small number of companies offering such a service. Burial and soil injection of pesticides have also been considered as means of disposal with sites being in areas where public health, terrestrial and aquatic systems will not be adversely affected. Sanborn et al. (1977) have studied the feasibility of disposing of large quantities of selected pesticides in the soil and have concluded that such disposal is feasible for EPTC, trifluralin, methoxychlor, carbaryl, and malathion but not for atrazine, paraquat, diuron, linuron, and endosulfan. Their general conclusion was that not enough information concerning degradation and metabolites exists to allow for concentrated soil disposal of most pesticides.

It is our belief that research concerning the disposal of pesticide containers should be encouraged. Techniques like the inoculation of soils with bacterial strains utilizing pesticide compounds have a potential application in both disposal and polluted areas. Changes in application equipment might also work to minimize the problems of losses due to improper filling and disposal techniques. One such method is the closed-system filling of spray tanks wherein the unopened pesticide container is inserted into an attachment on the sprayer. After the pesticide has flowed out of the container, the container is rinsed and punctured, ready for recycling or burial in approved areas. This system, which is mandatory in California, insures proper disposal of containers while reducing poisoning risks to the applicator. Tank rinsate disposal problems might also be solved by equipment changes. At present, a new application procedure is being developed which circumvents this problem. Instead of mixing an entire tank of dilute pesticide before application begins, new spray equipment mixes the pesticide concentrate with water as it comes through the spray nozzle. By using this technique farmers avoid the problem of disposing of dilute spray materials not used on the crop. At the end of spraying procedures, unused pesticides are in their original concentrated form, ready to be stored and eventually reused. Given the high cost of many pesticides (atrazine costs over $50 for five gallons), the farmer has an incentive to save the unused concentrate.

As the problem of disposal is similar for all pesticides, we will not discuss the significance of these losses specifically for the case study pesticides.

REPRESENTATIVE PESTICIDES

The chemical characteristics of pesticides themselves as well as in the methods of application and availability of alternative pest control programs vary so much that it is difficult to make general statements about the relative effectiveness of SWCP and other methods. Hence, we have selected five pesticides to examine in detail. These pesticides are representative of major classes of pesticides and thus allow tentative conclusions to be made concerning the importance of SWCPs for their close chemical relatives.

Pesticides move into the environment through runoff, drift, volatilization, and possibly leaching. Each of the methods we have discussed affects the amounts carried by each of these pathways in different ways. Improved pest control programs like IPM which reduce the amount of pesticide applied, result in lower amounts of pesticide being carried in all pathways. SWCPs in most cases, reduce only the amount carried in runoff. Adjustments in spray procedures or equipment are effective for reducing drift losses, but by increasing pesticide efficiency, could reduce the amount of pesticide applied to the field and thus, could reduce losses by runoff and leaching.

The following sections discuss for each of the representative pesticides the effectiveness of SWCPs and methods for increasing the efficiency of pesticide use. Since the methods have varying effects on the amounts carried in different pathways, the possible environmental impact associated with material in each pathway is discussed to the extent possible given the limited data available.

The benefits of each of the methods for pollution control are limited by the maximum amounts of pesticide carried in each of the transport pathways. Table 9-8 gives a summary of literature reports on amounts of each of the representative pesticides which were transported through runoff, drift and volatilization. For the foliar insecticides, toxaphene and methyl parathion, losses in runoff are insignificant in comparison with the amount of material lost in drift and volatilization. Very little carbofuran is carried by runoff, and that which is carried in runoff degrades very rapidly in aquatic systems. Paraquat losses in sediment are much higher than for drift and volatilization; however, as is discussed below, the paraquat carried in sediment does not appear to pose a serious environmental hazard.

Methyl Parathion

Methyl parathion is a broad spectrum organophosphate insecticide characterized by high toxicity and a low level of persistence. Rapid disappearance in soil limits methyl parathion's insecticidal control to those pests attacking the foliar portions of the crop, while rapid disappearance from treated plant surfaces necessitates repeated applications throughout the growing season. Eighty-four percent of the methyl parathion used in U.S. agriculture in 1971 was applied to cotton (Table 9-1) where it replaced the DDT formerly used in combination with toxaphene for the control of the boll weevil, and the bollworm complex.

Methyl Parathion: Runoff and Leaching--

The potential for runoff losses of methyl parathion from agricultural lands is limited by factors occurring both before and after the material reaches the soil environment. Substantial drift losses and volatilization and degradation from plant and soil surfaces reduce the amount of material available for runoff, while physical characteristics of the material limit its mobility in and from soil. Sheets *et al.* (1972) studied the runoff of methyl parathion from cotton in Rocky Mount and Lewiston, North Carolina, over a six-month period and found that losses ranged from 0.008 to 0.25 percent of the 13.4 kg/ha applied to the four plots studied.

Beyerlein and Donigian (Appendix F) have done the only modelling study on the runoff of methyl parathion from cotton agricultural lands. The model estimated losses of 4.6 percent of the material applied, with 4.2 percent of the losses occurring in the dissolved phase and 0.4 percent occurring in the solid phase of the runoff. These predictions are an order of magnitude higher than those measured in the empirical study mentioned above, and unfortunately could not be validated because of the lack of field data. The higher runoff noted in this study may be partially explained by the exclusion of drift losses and volatilization losses except as a function of degradation.

TABLE 9-8. PESTICIDE LOSSES AS A PERCENTAGE OF THE MATERIAL APPLIED

	Atrazine	Paraquat	Carbofuran
Chemical Class	s-Triazine	Bipyridylium	Carbamate
Type Application	broadcast or band applied by ground or air equipment	spray application by ground or air equipment	soil treatment by ground equipment Broadcast application of granules and foliar spray treatment by air or ground equipment
Application Rate (kg AI/hectare)	2.2 - 4.4	.27 - 1.1	1.68 - 3.36
Drift losses	negligible to 40%[1]	negligible to 40%[1]	negligible[2] (soil treatment) to 40% (foliar spray)
Volatilization losses	11%[5]	insignificant[6]	insignificant[7]
Runoff losses	average 5% (see Table 9-9)	1.28 - 22%[10]	0.5 - 2.0%[11]

TABLE 9-8. PESTICIDE LOSSES AS A PERCENTAGE OF THE MATERIAL APPLIED (Continued).

	Methyl Parathion	Toxaphene
Chemical Class	Organophosphate	Organochlorine
Type Application	foliar application by ground or air spray equipment	foliar application by ground or air spray equipment
Application Rate (kg AI/hectare)	0.13 - 3.36 (3-10 applications/ season)	1.1 - 3.36 (1-12 applications/ season)
Drift Losses	50 - 60%[3]	50%[4]
Volatilization	33%[8]	24%[9]
Runoff Losses	0.05 - 4.8%[12]	0.36%[13]

FOOTNOTES FOR TABLE 9-8.

[1] von Rumker *et al*. (1975). A general percentage for all herbicides based on application procedures.

negligible = broadcast surface application of spray by ground equipment.

40% = broadcast surface application of spray by air equipment based on estimated losses from U.S. corn crop. Baker and Johnson (1977) found that atrazine had 48-51% drift losses using ground equipment under windy conditions (windspeed 20 km/hour).

[2] von Rumker *et al*. (1975).

negligible = broadcast application of granules by ground equipment.

40% = foliar application of low-volume spray by air equipment. From table on likelihood of pesticide drift during crop treatment in agriculture by method of application.

[3] Adair *et al*. (1971). Test of aerial application at an altitude of 1.5 meters. 40-50% of applied pesticide found on ground in area which included the target field plus the area 800 meters downwind from the field.

[4] Ware *et al*. (1970). Studied drift of aerially applied toxaphene with both spray and dust formulations at wind speed of 4.8-6.4 km/hour. Actual amount of applied toxaphene reaching plants and the ground was 47.7% for the spray applications, 14% for the dust formulation.

[5] Foy (1963). 11% evaporated from the soil surface in two days at 40°C and an air velocity of 2 liters/minute.

[6] Calderbank and Slade (1976).

[7] von Rumker *et al*. (1974).

[8] Haque and Freed (1974), estimated volatilization losses from a loam soil at 25°C under annual rainfall of 150 cm = 7 - 14 kg/ha-year or more. Here we assume application rate 3.36 kg/ha, 10 times/season = 33.6 kg/ha-year, volatilization average = 11 kg/ha-year.

FOOTNOTES FOR TABLE 9-8 (Continued)

[9] Nash *et al*. (1977). Volatilization losses estimated in agroecosystem chamber where toxaphene was applied at a rate of 2.0 - 2.7 kg/ha to cotton plants at weekly intervals for 6 weeks.

[10] Baker *et al*. (1979). Found runoff losses 1.28 to 3.34%. Smith *et al*. (in press) found runoff losses as high as 22% on sediment.

[11] Caro *et al*. (1973). Study on two watersheds planted with maize.

Year	Amount Applied kg AI/ha	Type Application	% of Applied Carbofuran Lost
1971	5.31	broadcast and disced into 7.6 cm depth	0.9
1971	4.08	band application - in furrow treatment	0.5
1972	3.05	band application	1.9

[12] von Rumker *et al*. (1975). Estimated typical runoff losses of methyl parathion to be 0.05% of the applied material. Beyerlein and Donigian (Appendix F), in a 10-year simulation study based on data from Georgia, found losses of 4.8% in sediment and water runoff.

[13] Bradley *et al*. (1972). In a 6-month study on DDT and toxaphene runoff from cotton plots found toxaphene runoff losses of 0.36%, 75% of which was associated with the sediment fraction.

Furthermore, Beyerlein and Donigian included a rather long half-life of 45 days for methyl parathion located in soil; half-life ranges of three to 11 days have been given for methyl parathion by other sources (USEPA, 1975b). The inclusion of this extended half-life in the model will allow for prolonged periods of methyl parathion's exposure to raindrop splash and surface runoff, and will inevitably produce higher estimates of runoff losses. Therefore, given these limitations and the lack of field data for model validation, it seems reasonable to assume that the empirically observed losses of less than 0.25 percent more accurately approximate typical runoff losses of methyl parathion from cotton.

Very little is known about the leaching losses of methyl parathion from agricultural soils, although one can surmise that losses are limited by the same factors which limit runoff losses. Li and Fleck (1972) have reported the presence of small amounts of the material (in the parts per trillion range) in surface and subsurface drain effluents. Haque and Freed (1974) have estimated that leaching will move methyl parathion less than 20 centimeters through the soil profile under an annual rainfall of 150 centimeters.

Methyl Parathion: Air-Borne Losses--

Substantial air-borne losses of methyl parathion from target areas may occur both during and after application procedures (Table 9-8). Haque and Freed (1974) estimated annual volatilization losses of 7-14 kg/ha or more from a silt loam soil under an annual rainfall of 150 centimeters. Given an application rate of 3.3 kg/ha repeated ten times throughout the growing season, volatilization losses of methyl parathion would comprise approximately 33 percent of the applied material. Volatilization losses from plant surfaces probably occur at a more rapid rate, and can be so large as to reduce methyl parathion's half-life to one half hour in cotton growing regions of Mississippi (Quinby _et al_. 1958). Drift losses of methyl parathion are also quite large as much of this material is sprayed on the crop by air equipment. In one test of drift from an aerial application at an altitude of 1.52 meters, only 40 to 50 percent of the applied methyl parathion was found in a sample area which included the target field and the area 800 meters downwind from the crop (Adair _et al_. 1971). von Rumker _et al_. (1975) have reviewed the literature and predict similar drift losses for any insecticide applied by aerial equipment.

The environmental significance of methyl parathion's air-borne losses is not known, although there are indications that drift into areas of human habitation could have insidious effects on public health (Gershon and Shaw, 1961). Residues of methyl parathion found in air samples taken oven Alabama, Florida, and Mississippi (Stanley _et al_. 1971; Arthur _et al_. 1976) prove that such air-borne losses do not simply disappear but persist in the atmosphere until they are degraded or redeposited on the earth's surface with dust particles or rainfall. Studies establishing the concentrations of methyl parathion in rainfall over agricultural areas would help determine whether the redepositing of this material contributes significant amounts of methyl parathion to aquatic and terrestrial ecosystems.

Methyl Parathion: Toxicity--

Methyl parathion's extreme toxicity to aquatic insects, and moderate toxicity to fish and lower aquatic fauna (Table 9-5) have allowed its successful use for mosquito control in surface waters (USEPA 1975b) Apperson et al. (1976) reported no lasting effects on zooplankton when methyl parathion was applied to Clear Lake for gnat control, and concluded that. in this case, direct applications of the material to the lake "achieve no perceivable permanent alteration of the lake ecosystem." However. Eisler (1970) has found that the toxicity of methyl parathion to fish may be increased in aquatic systems having increased oxygen levels. Holden (1973) has further shown that sublethal concentrations of methyl parathion affect reproduction by inducing abortion in fish.

Effects on terrestrial non-target species appear to be of a more severe nature. von Rumker et al. (1974) report high levels of toxicity to soil fauna, although soil microflora may actually benefit from lower dosages of methyl parathion because of their ability to use the insecticide as an energy source (Naumann, 1970). Methyl parathion is highly toxic to insect predators and parasites (Wilkison et al. 1975; Edwards and Thompson, 1973) and has been responsible for numerous honey bee kills. Honey bee mortality has been especially high in those cases where methyl parathion was applied in a microencapsulated form to increase its residual activity (Burgett and Fisher, 1977). Methyl parathion is highly toxic to mammals and birds, and has caused bird kills in treated areas (U.S. Dept. of Interior, 1966; Bejer-Peterson et al. 1972).

Mammals exposed to sub-lethal doses of methyl parathion over a period of time will suffer from symptoms occurring as a result of progressive inhibition of cholinesterase (von Rumker et al. 1974). Gershon and Shaw (1961) studied the case histories of 16 people suffering from psychiatric disorders who had also been repeatedly exposed to organophosphate materials over periods of 1 1/2 to 10 years. In a follow-up of four cases, the authors noted that severe impairment of memory, difficulty in concentration and reactions of a schizophrenic and depressive nature persisted for six months after the cessation of exposure to organophosphate insecticides. Normal behavior was observed 12 months after exposure ceased.

The potential for biomagnification of methyl parathion is lessened by its low fat solubility, high toxicity, and rapid degradation in plants, animals, microorganisms, and aquatic systems (Sanborn et al. 1977; von Rumker et al. 1974; USEPA, 1975b). However, residues have been found in sea animals (USEPA, 1975b) and Apperson et al. (1976) have suggested a further investigation of the possible bioaccumulation of methyl parathion in bluegill and sunfish.

Methyl Parathion: Persistence--

Methyl parathion, like other organophosphates, is considered to be a relatively nonpersistent insecticide, degradable by nonbiological factors and biological organisms (Table 9-6). Though often more toxic than the parent compound, the by-products of these degradation pathways are not considered to be of great environmental importance as extreme instability causes their rapid disappearance from terrestrial and aquatic ecosystems (von Rumker et al. 1974). Methyl parathion has a short residual life on treated plants. In midsummer, applied concentrations of 106 ppm may fall to 3.9 ppm 96 hours after application (Ware et al. 1972). None of the studies conducted on methyl parathion's disappearance from treated plants stated whether the believed mechanism of dissipation was by degradation or volatilization. Degradation of the material entering the plant does not appear to be as rapid, for methyl parathion is commonly found in food and feed samples examined for residues (Duggan and Duggan, 1973).

The supposed nonpersistence of methyl parathion in aquatic systems has led some researchers to dismiss the possibility of serious contamination being caused by this material. However, a review of the literature indicates that methyl parathion residues are moderately persistent in aquatic ecosystems. Apperson et al. (1976) in their study on Clear Lake found that there was a carry-over of methyl parathion residues between insecticide treatments occurring at intervals of 20 days. The absence of residues in sediment samples suggested that degradation took place before or upon penetration of the bottom mud. Eichelburger and Lichtenberg (1971) found that methyl parathion degraded completely in bottled river water within eight weeks, and data from the USEPA (1975b) show persistence rates of less than one month in soil-water and less than four months in lake water.

Methyl parathion's degradation in soils generally occurs at a rapid rate, though the process can be slower when the material is applied at heavier dosages and in those areas where soils are alkaline or contain high proportions of dry sand or clay (Nayshteyn et al. 1973; Sanborn et al. 1977). Lichtenstein and Shulz (1964) applied 5.6 kg/ha of methyl parathion to a Carrington silt loam and found that only eight percent of the insecticide remained 15 days after treatment. Studies by King and McCarty (1968) estimating the theoretical half-life have shown similar results. Methyl parathion's persistence in soil has been reassessed in recent persistence studies using ^{14}carbon labelled insecticides (Katan et al. 1976; Lichtenstein et al. 1977). In 1977, Lichtenstein et al. published data showing that only seven percent of the applied methyl parathion was extractable 28 days after soil treatment, with an additional 43 percent existing in the form of non-extractable bound residues.

Questions raised about the nature and potential toxicity of these bound residues resulted in a study by Fuhremann and Lichtenstein (1978) which investigated the release and potential pickup of the residues by earthworms and oat plants. Data showed that bound ^{14}carbon residues of methyl parathion were incorporated into the tissues of earthworms and oat plants exposed to the soil-sand-bound residue mixture. In the case of the earthworms, 58 to 66 percent of the adsorbed ^{14}carbon residue was converted to a strongly bound form within the bodies of the worms. The remainder of the ^{14}carbon residues was converted to a soluble form. In the case of the oat plants, 82 to 95 percent of the ^{14}carbon residues were extractable as benzene or water soluble ^{14}carbon compounds. The authors concluded that bound residues are not excluded from environmental interactions, and suggested that further research be done to determine the importance of soil-bound residues in biological systems.

Methyl Parathion: The Effectiveness of SWCPs in Reducing Transport--

Methyl parathion's small runoff losses coupled with its relatively rapid degradation in water and moderate toxicity to aquatic organisms indicate that only minimal benefits will be gained if SWCPs are implemented to control pollution resulting from the use of methyl parathion. The only results which might temper these conclusions are given by Beyerlein and Donigian in their simulation study on methyl parathion (Appendix F). Their model predicts that methyl parathion's losses in runoff would comprise 4.6 percent, 3.9 percent, 3.8 percent, and 2.3 percent of the amount originally applied under conventional tillage, minimum tillage, contours, and contours with terraces, respectively. Thus, according to this model, runoff losses of methyl parathion could be reduced by 50 percent using contours with terraces. If one uses the larger runoff losses obtained by Beyerlein and Donigian, 50 percent reductions in runoff will prevent 156 g/ha from leaving cotton fields; if one uses the field results of Sheets *et al.* (1972), 50 percent reductions will only prevent 0.55 to 16.5 g/ha from leaving cotton fields. As was pointed out earlier, Beyerlein and Donigian did not include air-borne losses and were not able to validate their model with field data. Without additional empirical information to substantiate these much higher runoff losses, it is difficult to build a strong case for the benefits of SWCPs for controlling methyl parathion's runoff losses.

Methyl Parathion: Increasing Efficiency in Use--

The substantial drift losses of methyl parathion (50 to 60 percent) noted in Table 9-8 indicate the need for increasing the accuracy of application procedures. Available information indicates that these drift losses could be substantially reduced by switching from aerial application to application with ground equipment. Although no field studies have been done on drift losses of methyl parathion with ground applications, a general report on pesticide drift losses indicates that drift losses can be reduced from an average of 40 percent of the material to five percent if foliar applications are made with ground equipment instead of aerial equipment (von Rumker *et al.* 1975). Ground applications thus have the advantage of increasing the accuracy of pesticide placement within the target area. In Texas, cotton field entomologist Ray Frisbie reported that ground applications cost about the same as aerial applications (within ± $2.50/hectare) and that, in most cases, the advantages of increased accuracy obtained with ground applications outweigh the advantages of aerial applications (personal communication, 1979). However, in some areas such a substitution of application procedures may be difficult or impossible to implement as aerial applications have the advantage of timeliness. When and if insect infestations are noted in a field, aerial equipment can be used to apply pesticides to large acreages in a short period of time. Ground applications of pesticides take more time and are hindered by muddy ground conditions resulting from heavy rain or irrigation. When and if such conditions make it impossible to substitute ground equipment for aerial application, it may be possible to reduce drift losses to non-target areas by making changes in the aerial equipment itself. Indeed, Ware *et al.* (1975) have shown that the use of Raindrop^{TM} nozzles can reduce off-target drift by 50 percent when compared to type D flooding nozzles.

In the case of methyl parathion, which is used primarily in cotton production (Table 9-1), integrated pest management programs represent a more comprehensive approach to increased efficiency of use and reduced losses to non-target areas. The survey by Pimentel et al. (in press) discussed in Table 9-7 indicates that if IPM techniques were implemented in all cotton producing regions where they are cost-effective (i.e., where IPM causes no net reduction in farm income), the national use of organophosphate insecticides in cotton production would be reduced by 46 percent. Casey et al. (1975) have studied the impact of cotton IPM programs on the use of methyl parathion and other insecticides in Texas, and reported that methyl parathion use dropped from 7.49 kg/ha to 0.24 kg/ha in the Trinity River region. The reduction in use was not as substantial in the Brazos River region (10.55 kg/ha to 5.38 kg/ha) but included 4.57 kg/ha used for a fall diapause program. Casey et al. estimated that if the new pest management strategy was implemented for all cotton hectares in the Trinity River and Brazos River regions, methyl parathion's annual use would be reduced from 849,724 kg to 355,678 kg. Such substantial reductions in pesticide use will inevitably reduce the amounts of methyl parathion reaching non-target areas via air-borne, runoff, and leaching losses. Within agricultural areas, reduced methyl parathion use will cause increased survival of beneficial insects, and fewer opportunities for improper disposal of pesticide containers and tank rinsate. In addition to the economic and environmental benefits gained by reductions in methyl parathion's use, Lacewell et al. (1976) have found that the cotton production system studied by Casey et al. (1975) required 56 percent less energy (kcal) to produce a kilogram of lint.

Methyl Parathion: Summary and Conclusions--

A review of the literature indicates that the potential for reducing transport of methyl parathion is much greater with methods which increase efficiency of use than with SWCPs. Empirical studies indicate that less than 0.26 percent of the methyl parathion applied to cotton leaves the field in runoff. Terraces with contours appear to be the most effective SWCP for the control of this runoff and reduce losses from 0.2 percent of the material applied to 0.13 percent. Drift of methyl parathion with commonly used aerial application procedures may produce pesticide losses to non-target areas which are over a hundred times larger than losses in runoff. These drift losses can be reduced substantially by switching from aerial to ground applications of the material. Information on the changes in costs and pest control effectiveness associated with a switch to ground applications is not available for most cotton producing regions. However, pest control experts in the cotton growing regions of Texas have indicated that such a change is not only feasible economically but is also desirable in terms of improving the effectiveness of control practices.

It has been argued that the methyl parathion carried in drift is less environmentally hazardous than the methyl parathion carried in runoff and that runoff control is therefore more important than drift control. A review of the literature does not support this conjecture. Methyl parathion is generally more toxic to terrestrial organisms and may have insidious effects on human mental health. Toxic effects will also be extended to aquatic species as a certain percent of the drifting material will settle on lakes, rivers, streams and drainage ditches. The lower toxicity of methyl parathion to aquatic species is well illustrated by the fact that the material has been registered for application directly to surface waters for mosquito control.

The most promising strategy currently available for reducing the transport of methyl parathion away from target areas involves improvements in the timing of spray operations and the use of biological and cultural control methods to supplement the pest control obtained with pesticides. Integrated pest management programs in cotton have reduced methyl parathion use by approximately 40 percent at no net cost to the farmer and have resulted in substantial reductions in energy use. This reduction in use would be expected to result in similar reductions in runoff, drift, and volatilization losses. Hence the impact of IPM programs on runoff losses is considerably greater than the impact of the less expensive SWCPs such as conservation tillage and contour plowing. The more expensive but most effective SWCP, terraces with contours, will reduce runoff losses as much as IPM programs, but it does not have the added benefits of reduced energy use, reduced drift and volatilization losses, and reduced hazards to farmworkers and non-target species living in agricultural areas.

SWCPs which have been implemented for erosion control can have a small beneficial effect in reducing the runoff of methyl parathion. As long as SWCPs do not require an increased rate of methyl parathion application, it does not appear that the implementation of SWCPs will increase the movement of methyl parathion into the environment. In addition, most SWCPs can be used in conjunction with other methods which increase the efficiency of pesticide use. However, if a decision must be made regarding the allocation of limited resources specifically for the reduction of methyl parathion pollution, currently available information indicates that SWCP should be given a lower priority than increasing the efficiency of pesticide use by wider implementation of IPM and drift control programs.

Toxaphene

Pollock and Kilgore (1978), in their extensive review of the literature concerning toxaphene, have stated that "toxaphene remains the most heavily used and perhaps, the least understood insecticide presently available." Although the technical product has been shown to contain nearly 200 polychlorinated camphenes (Holmstead *et al*. 1974), its chemical composition remains largely undetermined. The lack of information about the metabolites and photoproducts of the numerous compounds making up toxaphene makes it difficult to ascertain the environmental hazards associated with its use.

Toxaphene: Runoff and Leaching--

Runoff losses of toxaphene apparently can occur but are limited by drift and volatilization losses which occur before the material is exposed to raindrop splash and surface runoff. The potential for runoff losses is further reduced by toxaphene's low solubility in water and strong attraction for soil particles. These characteristics place toxaphene in the solid phase of runoff and thus limit its transport to the small amounts of sediment found in runoff leaving agricultural fields.

Field studies on losses of toxaphene in runoff support these conjectures and report losses which make up only a small percent of the applied material. Bradley et al. (1972) studied the runoff of toxaphene from cotton in North Carolina and found that losses ranged from 0.078 to 0.72 percent of the 26.9 kg/ha applied to the eight plots under observation. Of this material, 75 percent was associated with the solid phase, with the remainder found in the dissolved phase of the runoff. Willis et al. (1976) have obtained similar results in their study of toxaphene runoff from larger cotton plots in Mississippi.

While these losses appear to be small when seen as a percentage of the material applied, they take on greater significance when it is observed that 0.078 to 0.72 percent of the 26.9 kg/ha applied results in losses of 21 to 200 g/ha of toxaphene. Given large tracts of cotton acreage, these losses could result in serious contamination problems in surface waters located in cotton growing areas. Bradley et al. (1972) did find that significant concentrations of toxaphene appeared in a 0.2 hectare pond located in the experimental watershed. Concentrations increased from levels of less than 1 ppb before the cotton was treated to 65 ppb at midseason, and equaled or exceeded the 96 hour median tolerance for bluegill on all but one occasion during the growing season. The authors attributed this toxaphene pollution to runoff but did not study possible residue inputs from drift losses and contaminated rainfall.

Leaching losses of toxaphene through agricultural soils are limited by the same factors limiting runoff losses. Haque and Freed (1974) estimated from the best available information that toxaphene will move less than 10 cm through the soil profile with an annual rainfall of 150 centimeters. Field studies have indicated a slightly higher propensity for leaching. Swoboda et al. (1971) found that after ten years, 90 to 95 percent of the toxaphene recovered from a Houston black clay was concentrated in the top 30 centimeters of the soil. LaFleur et al. (1973) applied 100 kg/ha of toxaphene directly to the soil surface to study the leaching behavior of "sterilant concentrations" of the material. In two months, toxaphene leached through approximately 300 cm of soil, and entered and contaminated groundwater for the entire year of observation. The direct application to soil, and the high rates of toxaphene used in the experiment are dissimilar to conditions of toxaphene use in the field, but may indicate the leaching hazards associated with spills or disposal of the material.

Toxaphene: Air-borne Losses--

Applications of toxaphene are predominantly made by aerial equipment, and can thus result in substantial drift losses to non-target areas (Table 9-8). Ware et al. (1970) studied drift losses of aerially applied toxaphene in windspeeds of 5.6 km/hr and found that only 47 percent of the spray-formulated material and 14 percent of the dust-formulated material were deposited on cotton plants. Gerhadt and Witt (1965) found similar increases in drift losses with dust formulations and reported that aerial applications of dust-formulated toxaphene produced downwind drift deposits which were six times greater than those obtained with spray formulations. Dust formulations of toxaphene have been largely abandoned because of associated drift losses.

Losses of toxaphene to the atmosphere continue to occur after the material has reached the soil surface and treated crops. Nash et al. (1977) found that 24 percent of the applied toxaphene volatilized from soil and cotton plant surfaces in a model ecosystem. The authors suggested that volatilization losses in the field might be larger because cotton plants are more widely spaced, and thus allow more of the volatilization losses from the soil to escape to the atmosphere. Swoboda et al. (1971) observed the rapid disappearance of toxaphene from a heavy clay soil and attributed it to large volatilization losses encouraged by daily soil temperatures of 60°C. Haque and Freed (1974) used available information to quantify annual vaporization losses of pesticides from soil, and estimated that 7 to 14 kg/ha of toxaphene would vaporize from a loam soil at 25°C.

The ecological importance of these atmospheric pollutants has not been ascertained, although air-borne toxaphene residues have been shown to have a half-life of approximately 15.1 days (Nash et al. 1977). Residue levels of 280, 170, 44, 86, and 220 ppt in five out of eight rainwater samples collected in Maryland indicate that some of the air-borne toxaphene is redeposited on the earth's surface (Munson, 1976). As mentioned previously, Goldberg et al. (1971) have suggested that aerial fallout such as this may greatly exceed river discharge in its contributions of pesticides to the oceans of the world. Toxaphene in rainfall and aerial fallout may also contribute to the agricultural soils. Further studies should be undertaken to determine how much toxaphene is actually being deposited by rainfall and aerial transport on aquatic and terrestrial ecosystems.

Toxaphene: Toxicity--

In general, there seems to be a serious lack of data concerning the effects of toxaphene on aquatic flora and fauna. In aquatic systems, isolation of the toxic component, or toxic group of components, is complicated by a lack of information concerning changes in the components' relative concentrations over time as a result of chemical reaction, microbial degradation, and preferential sorption by flora and fauna (Veith and Lee, 1971). However, whatever the mechanism or components of toxicity may be, toxaphene in toto is extremely toxic to lower aquatic organisms, having inhibitory effects on algae, plankton, and protozoa at concentrations as low as 0.1 ppm (Thompson, 1973; Butler, 1977; Sanborn et al. 1977). Smaller concentrations (1 ppb), while not toxic themselves, will have synergistic sublethal effects on oysters when in the presence of 1 ppb of both DDT and parathion (Lowe et al. 1970).

Although some fish species have developed resistance, toxaphene is in most cases extremely toxic to fish (Table 9-5), causing mortality at concentrations in parts per billion (Sanborn et al. 1977). The hazards posed by this degree of toxicity are prolonged by persistence rates of up to 12 months in water (Edwards, 1977) and have been responsible for numerous fish kills, and a general decline in the diversity of aquatic ecosystems located in cotton producing areas (Johnson, 1968; Grzenda et al. 1964; Keiser et al. 1973; Reimold, 1974; Reimold et al. 1973). Mehrle and Mayer (1977) have reviewed the literature concerning the sublethal effects of pesticides on fish and found that toxaphene inhibits collagen synthesis thereby depressing bone development in young fish. Other studies have indicated that sublethal concentrations of toxaphene may cause behavioral changes which jeopardize the survival of fish by affecting their ability to learn or react properly to external stimuli (Holden, 1973; Warner et al. 1966).

Less is known about the effects of toxaphene on terrestrial organisms. Martin et al. (1959) have shown that high levels of agricultural use of toxaphene can inhibit bacteria and fungi populations in soil. Normal dosages of toxaphene are highly toxic to insects including insect parasitoids and predators (Wilkison et al. 1975); however in some areas, target insects have developed resistance. Harris (1972) noted that the LD_{50} for a toxaphene-DDT mixture (2:1) varied throughout the state of Mississippi and ranged from 10 to 21 μg per larvae for the bollworm, and from 34 to 1000 μg per larvae for the budworm. Toxaphene is of intermediate toxicity to birds and mammals (Pollack and Kilgore, 1978) but has been responsible for two bird kills reported by Rudd (1966) and Keith (1966). While it has been reported that mammals metabolize most of the discovered components and that birds do not seem to accumulate residues in their fat tissue, not all of toxaphene's fat soluble metabolites have been identified, and analytical standards have not been available for those components which have been identified (Casida et al. 1974; Holden, 1973).

Toxaphene is similar to other organochlorines in that it is both stable and highly fat soluble, two conditions conducive to biomagnification. A number of studies have indicated that biomagnification of toxaphene does occur, although generally at lower concentrations, and with lower persistence than most other organochlorine insecticides (Guyer et al. 1971). Sanborn et al. (1976) measured ^{14}carbon toxaphene biomagnification in an aquatic-terrestrial model ecosystem, and found that concentrations in algae, snails, mosquito, and fish were 6902, 9600, 890, and 4247 times higher than the concentration found in the water, respectively.

Toxaphene: Persistence--

Like most organochlorines, toxaphene has a degradation rate measured in years (Table 9-6) rather than the weeks and months which describe the degradation rates of other pesticides. Although it is known that toxaphene is degraded by biological as well as non-biological factors (von Rumker et al. 1974), analytical problems arising from its complex structure complicate any assay of degradation rates and pathways. Toxaphene's persistence on plants is greatly reduced by volatilization losses to the atmosphere. The effect of wind on volatilization losses has been shown by Mistric and Gaines (1953) who found that toxaphene residues had 85 percent reductions in effectiveness when subjected to a simulated wind, with nine percent reductions seen in the absence of the wind. When sprayed on kale, most of the insecticide disappeared within three days without precipitation (Klein and Link, 1967). The toxaphene which escapes volatilization through adsorption into plant tissues, persists at low levels as a contaminant in cotton and various root crops (Nash, 1974).

Toxaphene's strong adherence to cotton leaves, and large volatilization losses from plant and soil surfaces allow only small amounts of the insecticide to reach or remain on the soil surface. The degradation of remaining residues is generally quite slow, but depends on the pH, temperature, and organic matter content of soils (Fowkes et al. 1960; Weed and Weber, 1974). Nash and Harris (1973) studied the persistence of toxaphene on a Congaree sandy loam, and found that 49 percent of the material remained 16 years after application. Randolph et al. (1960) found that 50 percent of the applied material degraded in one year in soil conditions more conducive to degradation processes.

Studies on toxaphene's behavior in surface waters report instances of rapid disappearance on the one hand and persistence of up to nine years on the other (Guyer et al. 1971). Some of these discrepancies may be due to toxaphene's chemical complexity and reversible sorption to aquatic flora and fauna (Pollock and Kilgore, 1978). Williams and Bidleman (1978) studied the degradation of toxaphene in an anaerobic salt marsh and found that residues were quickly degraded in sterile and non-sterile sediment to compounds having gas chromatographic retention times shorter than those of standard toxaphene components. Lee et al. (1977) found "weathered" toxaphene components to be less toxic to bluegills and concluded that some of the more toxic components had indeed degraded.

Several mechanisms for the "disappearance" of toxaphene from aquatic systems have been suggested. MacKay and Wolkoff (1973) predicted high evaporation rates and resultant short half-lives in water for the organochlorines because of their high activity coefficients while in solution and concluded that "evaporation from rivers and lakes may thus represent a major mechanism of the transport of pollutants through the environment." Hughes and Lee (1968) noted the great adsorption capacity of suspended sediments and suggested that sorption activities might play a significant role in the removal of toxaphene from natural water systems. A further evaluation of the detoxification role of lake sediments was made by Veith and Lee (1971) who found that toxaphene residue concentrations in the top five centimeters of the sediment increased for 190 days after application and then decreased by a factor of two every four months. Leaching of residues through the aquatic sediment did not appear to contribute to this disappearance as residues were not found at levels lower than 20 centimeters. This discovery of the detoxification role of sediments cast light on several older studies which showed that detoxification occurred with greater ease in well-mixed, shallow lakes (Terriere et al. 1966) and in surface waters containing large amounts of suspended sediments (Stringer and McMynn, 1958).

Veith and Lee (1971) have shown that the adsorption of toxaphene residues to suspended and settled sediment is irreversible. However, Reimold and Durant (1974) have shown that these apparently irreversibly adsorbed residues are not excluded from biological interactions. Their study of the pesticide residue levels resulting from the dredging of a toxaphene contaminated stream showed concentrations of 200 ppm in mummichog (Endulus heteroclitus) and 7.5 ppm in salt marsh cordgrass. The toxaphene found in the marsh grass was apparently translocated by the plant from contaminated sediments into plant tissue.

Toxaphene's persistent nature and heavy use in agriculture would lead one to expect numerous reported incidents of residues in plants, animals, soil and water; however this generally has not been the case. Toxaphene is rarely found in water samples (Pollack and Kilgore, 1978) although a few incidents of pollution have been reported. Barthel et al. (1969) have reported that toxaphene is a contaminant of the lower Mississippi and its tributaries. Nicholson et al. (1964) found toxaphene in all samples taken from a stream draining a 1000 square kilometer watershed which contained approximately 6060 hectares of cotton, and they reported toxaphene residues in all water samples taken before and after treatment by a water-treatment plant. Traces of two heptachloronorbornenes, compounds which could be degradation products of toxaphene, have been found in New Orleans drinking water at levels of 0.04 to 0.06 ppb (USEPA, 1974). Toxaphene has also been found in rainwater samples in Maryland (Munson, 1976) and in air samples taken over Mississippi (Stanley et al. 1971; Arthur et al. 1976) and areas as far away as Bermuda (Bidleman and Olney, 1975).

The 1971 National Soil Monitoring program sampled soils in 37 states and found toxaphene residues at only 6.2 percent of the sites sampled (Carey et al. 1974). However, the national results of this survey are somewhat deceiving since cotton-growing states such as South Carolina, Louisiana, and Mississippi had toxaphene residues in 76.5 percent, 30.8 percent and 71 percent, respectively, of the soil samples tested. Analytical techniques insensitive to toxaphene concentrations lower than 10 ppm undoubtedly also contributed to the small percent of toxaphene contamination reported in the national results.

Pollock and Kilgore (1978) reviewed the small amount of data that have been gathered on toxaphene's persistence and residues and concluded that the paucity of pollution reports resulted from inadequate analytical techniques and potential interference from other organochlorine insecticides present in monitoring samples. Pollock and Kilgore (1978) predicted greater success in uncovering toxaphene residues in future years because of better analytical methods, greater awareness of chemical analysts, and the declining use of other organochlorines whose presence interfered with attempts to monitor toxaphene.

Toxaphene: The Effectiveness of SWCPs in Reducing Transport--

It is quite difficult to evaluate the effectiveness of SWCPs in reducing toxaphene losses to non-target areas. On the one hand, persistent toxaphene residues seem to be ubiquitous in the soils of cotton-growing areas and thus have the potential for entering surface waters via runoff. On the other hand, runoff losses are dwarfed by drift and volatilization losses occurring during and after the application of toxaphene to the crop. The contributions of these air-borne losses to surface water pollution are not known, thus SWCPs could be implemented only to find that a good deal of the contamination resulted from the fallout of air-borne losses. Furthermore, Lichtenstein et al. (1971) found that the organochlorines, aldrin and heptachlor and their metabolites persisted for longer periods in fields with continuous crop cover than in fields which were annually cultivated. Thus, it is possible that SWCPs involving reduced cultivation may similarly prolong toxaphene's persistence in soil by limiting volatilization losses and degradation processes. This prolonged persistence could counter the benefits gained by SWCPs by extending the period of toxaphene's exposure to raindrop splash and surface runoff. As discussed in Section 7, reduced persistence of a pesticide may have a greater impact than any SWCP in reducing runoff losses of pesticides from agricultural lands. Hence, in some cases, prolonged persistence may counter the reductions in runoff gained by the implementation of certain non-structural SWCPs.

An evaluation of the effectiveness of SWCPs for reducing toxaphene's losses is further limited by the absence of studies determining the effects of various SWCPs on the runoff losses of toxaphene. As toxaphene does travel primarily with the sediment suspended in runoff (75 percent in the solid phase), one can extrapolate/estimate the effects of SWCPs by looking at another strongly adsorbed pesticide, paraquat. Beyerlein and Donigian's modelling results (Appendix F) indicate that SWCPs reduce paraquat losses by a maximum of 52 percent. One would not expect toxaphene losses in runoff to be reduced by more than 52 percent which amounts to approximately 78 g/ha.

Toxaphene: Increasing Efficiency in Use--

Since the primary use of toxaphene is in combination with methyl parathion for the control of cotton insect-pests, programs to increase the efficiency of its use are similar to those mentioned in the section on methyl parathion. Drift losses are large as the material is mostly applied by aerial equipment. Reductions in these losses can be made either by changing to application by ground equipment or by improving the accuracy and placement of toxaphene by aerial equipment. As in the case of methyl parathion, integrated pest management programs have allowed farmers to reduce the number of toxaphene treatments needed for the control of the insect-pest complex of cotton. Pimentel et al. (in press) interviewed numerous entomologists throughout cotton growing regions in 1974 and found that the use of organochlorines would have been reduced by 39 percent if IPM techniques available at that time had been implemented in all areas where they were cost effective. Casey et al. (1975) monitored the impact of an IPM program on pesticide use in the Brazos River and Trinity River regions of Texas and found that the traditional annual applications of toxaphene (1.41 kg/ha and 1.91 kg/ha) were entirely eliminated. The authors estimated that if this IPM program could have been implemented on all cotton hectares in the Trinity River and Brazos River regions, 133,367 kilograms of toxaphene would have been eliminated from cotton production. This elimination of toxaphene use was achieved without increases in the use of other insecticides and without extra cost to farmers. In fact, net returns on a per hectare basis increased by $55.87 in the Brazos River region and by $85.50 in the Trinity River region. Annual insecticide use on a per hectare basis was reduced from 14.51 kg/ha to 7.19 kg/ha in the Brazos River region and from 12.28 kg/ha to 6.22 kg/ha in the Trinity River Region. Such IPM programs thus provide the ultimate means of reducing any pollution associated with toxaphene's use in cotton production.

Toxaphene: Summary and Conclusions--

Both air-borne and runoff losses of toxaphene can contribute to the pollution of surface waters; however, approximately three-fourths of the toxaphene carried in runoff is adsorbed to suspended sediment and is thus not immediately available to biological organisms. Air-borne losses settled on surface water and dissolved toxaphene in runoff are readily available to biological organisms and thus have an immediate impact on aquatic ecosystems. Currently available information indicates that the amounts of toxaphene transported by drift and volatilization average over 60 percent of the material applied (52% + 0.24 [47%]). Hence air-borne losses of toxaphene are two orders of magnitude greater than the amounts of toxaphene transported in runoff. Unfortunately, an evaluation of the relative impact of air-borne and runoff losses on water quality is not possible because studies have not been done to measure actual imputs of air-borne and runoff pollutants to surface waters. However, given the much larger air-borne losses and the fact that most toxaphene carried with runoff is adsorbed to sediment, it is reasonable to assume that air-borne losses of toxaphene are at least as hazardous to aquatic systems as the toxaphene contributed by runoff.

In the absence of field or modelling studies on the impact of SWCPs on the runoff of toxaphene, evaluations of the effectiveness of SWCPs for the control of toxaphene pollution must rely on extrapolations from data on the effectiveness of SWCPs in controlling the runoff of other pesticides from agricultural fields. Given the results for other related pesticides, it is unlikely that any SWCP would reduce toxaphene transport in runoff by more than 50 percent. Empirical studies have generally found no more than 0.5 percent of the applied toxaphene in runoff, thus the maximum impact of SWCPs on toxaphene's transport would involve a reduction of only 0.25 percent of the material applied.

The substantial drift losses which occur with the most commonly used application procedures (53 percent) can be reduced significantly by changes in the aerial equipment itself or by changes from aerial applications to ground applications. Little published information exists to indicate whether in all cotton growing areas such a substitution is economically feasible or possible in terms of the goals of pest management. However, pest management specialists in Texas have indicated that this substitution is not only feasible economically, but also desirable in that drift losses significantly reduce the effectiveness of expensive pesticide treatments.

Increasing efficiency of pesticide use through the implementation of integrated pest management programs is a widely available, inexpensive means of substantially reducing the amount of toxaphene reaching non-target areas. The results of a national survey indicated that IPM would reduce organochlorine use by about 39 percent using currently available methods and by 59 percent using methods which will be widely available in the next five years. Hence, the impact of currently available IPM programs on toxaphene's runoff losses would probably be almost as great as any of the SWCPs. The difference in runoff losses between the most effective SWCP studied are available IPM programs is on the order of only 0.05 percent of the material applied ([0.5-0.4][0.5%]). IPM methods available in the near future would be expected to be more effective than SWCPs in reducing runoff losses. IPM programs have additional advantages over SWCPs in that they have allowed farmers to cut down on crop production costs and energy needs while reducing toxaphene's drift and volatilization losses. These reductions are substantial, amounting to approximately 25 percent of the material applied to cotton. It should be noted that reducing application rates is the only feasible way of decreasing the sizeable volatilization losses of toxaphene.

In conclusion, SWCPs, changes in application procedures, and IPM are all effective means of reducing toxaphene's transport away from target areas. However, IPM programs combine reductions in runoff and air-borne losses with reductions in energy needs and crop production costs and thus appear to be the most cost effective means of reducing toxaphene contamination in both target and non-target areas. IPM programs should therefore be given a higher priority than SWCPs or changes in application procedures when allocating limited resources for the control of toxaphene pollution. However, given the large amounts of air-borne losses, changes in application procedures should be encouraged in conjunction with IPM programs where feasible.

Carbofuran

Concern about the environmental hazards associated with the use of persistent pesticides encouraged the development of several insecticides of a nonpersistent nature which are now considered to be safer materials in spite of their greater toxicity to non-target animals. One of these recently introduced insecticides is carbofuran, a broad spectrum contact and systemic carbamate which has been especially successful in granular, soil treatments for the control of corn rootworm. Carbofuran is applied as a foliar spray to a much lesser degree and can be used to control various insect pests of alfalfa, corn, peanuts, potatoes, rice, tobacco, peppers, and sugarcane.

Carbofuran: Runoff and Leaching--

Although carbofuran's physical and chemical characteristics indicate a strong potential for runoff losses, empirical studies have generally reported losses of less than two percent of the applied material. Unusually large single event losses of up to 14 percent have been reported in areas with claypan soils and large losses of runoff water. After reviewing studies on runoff, von Rumker *et al*. (1975) estimated that runoff losses will comprise no more than one percent of the carbofuran applied to the U.S. corn crop.

Caro *et al*. (1973) studied carbofuran in runoff from a silty loam soil with soil treatments by broadcast and in-furrow applications and reported losses of 0.5 to 2.0 percent of the original material. Although the carbofuran applied as an in-furrow treatment persisted for a longer period of time (117 days versus 46 days for the broadcast material), runoff losses were less for a given volume of runoff when in-furrow applications were used. Runoff losses were transported primarily in the dissolved phase of the runoff and were higher after the granules had dissolved and released the active ingredient. Smith *et al*. (1974) have studied loss of carbofuran from conventionally tilled corn grown on claypan soils and found larger runoff losses than those reported from silty loam soils. Carbofuran was applied at a rate of 1.12 kg AI/ha and suffered single event runoff losses which ranged from zero to 10.73 g/ha in 1971, 32.4 to 56.6 g/ha in 1972, and zero to 161.5 g/ha in 1973. Maximum single event losses in each year in terms of a percentage of the material applied consisted of 0.96 percent in 1971, 5.05 percent in 1972, and 14.42 percent in 1973; maximum concentrations in water ranged from 298 to 600 ppb in the three years of the study.

The FMC Corporation also conducted runoff studies (reported in USEPA, 1976), and found carbofuran residues of 1 ppm in a neighboring pond when heavy rainfall followed four days after corn treatment with 4.48 kg AI/ha. This contamination was reduced to "negligible" concentrations by the time the next sample was taken 16 days later. As the 96-hour LC_{50} for ten fish species range from 0.08 to 1.18 ppm (Pimentel, 1971), these runoff losses constitute a serious threat to non-target species.

Leaching losses are limited although carbofuran's physical and chemical characteristics might lead one to expect otherwise. Leaching studies conducted by the FMC Corporation (USEPA, 1976) in Nebraska and Georgia show that the carbofuran remained primarily in the top 15 centimeters of soils rich in clay and organic matter. Rapid degradation in soils will in most cases limit both runoff and leaching losses of carbofuran from treated fields.

Carbofuran: Airborne Losses--

The studies thus far conducted on carbofuran's air-borne losses from agricultural fields indicate that losses are generally quite small, since the material is primarily applied as granules for soil treatment (Table 9-8). von Rumker *et al.* (1975) reviewed the data on drift losses and estimated drift losses to be much larger with foliar spray treatments than with granular treatments of the soil by air or ground equipment. Drift losses of foliar spray treatments can range from one to ten percent with application by a tractor boom sprayer, to 10 to 60 percent with application by an aircraft boom sprayer; losses with granular treatments are negligible because most granules are made to be 250 μm or greater in size. It is possible that larger drift losses occur when granules are applied in windy conditions. Baker *et al.* (1977) applied 1.12 kg/ha of fonofos as granules in 20 km/hr winds on three different occasions and found only 0.84 kg/ha, 0.63 kg/ha, and 0.45 kg/ha of fonofos in the soil immediately after application. Losses to the atmosphere through volatilization from treated soils and crops are small or perhaps nonexistent (von Rumker *et al.* 1974). Harris and Miles (1975) classified carbofuran as a non-volatile insecticide based on measurements of fumigant activity against insects (Table 9-8).

Carbofuran: Toxicity--

Carbofuran is quite toxic to aquatic organisms at low concentrations. Sangha (1972) and Yu *et al.* (1974) studied carbofuran's effect on lower aquatic organisms in a model ecosystem and found that fresh water clams, fresh water crabs, frogs, snails and water fleas were killed when sorghum plants in the terrestial part of the ecosystem were treated with a normal application rate of 1.12 kg AI/ha. In spite of carbofuran's observed rapid hydrolysis in water (Yu et al. 1974), successful re-establishment of test organisms did not occur until the twentieth day after carbofuran's application. Carbofuran is also quite toxic to fish and produces mortality in 50 percent of the population for ten fish species after 96 hours of exposure to concentrations of 0.08 to 1.18 ppm (USEPA, 1976). Sanborn (1974) used the model ecosystem to construct a food chain which included sorghum, frogs, saltmarsh caterpillar larvae, snails, freshwater clams, freshwater crabs, water fleas, green filamentous algae, mosquito larvae, and mosquito fish. At the end of 33 days, organisms were analysed for ^{14}C labelled carbofuran; carbofuran did not accumulate in any of the test organisms, and Sanborn concluded that the material was highly biodegradable and posed no hazards in terms of biomagnification.

Carbofuran is a direct inhibitor of acetylcholinesterase and thus exhibits high toxicity to both avian and mammalian species (Table 9-5). Toxic effects on birds have resulted in numerous bird kills (USEPA, 1976), one of which was cited as perhaps the worst single incident of a wildlife kill involving pesticides in the United States. In this particular incident, 2400 wild ducks died after exposure to the carbofuran which had been applied for the control of Egyptian alfalfa weevil and pea aphids (Fukuto, 1976). This high level of toxicity greatly limits carbofuran's use as an insecticide, and has prompted the development of related carbamate insecticides with lower degrees of mammalian and avian toxicity. Bioaccumulation of carbofuran has not been noted in birds or mammals and is deemed highly unlikely because of the insecticide's rapid degradation in biological systems, and low fat solubility (USEPA, 1976).

Carbofuran's toxic effects on beneficial anthropods are more severe with foliar spray applications than they are with granular soil applications. Bee kills and significant decreases in earthworm populations have been observed when these organisms came into contact with the flowable formulation (Tomlin, 1975; USEPA, 1976). Soil incorporation of carbofuran should reduce toxic effects on various insect predators in many cases; however, Brown and Shanks (1976) have shown that populations of two predators of the two spotted mite on carbofuran-treated plants showed significant mortality. Since carbofuran is a systemic insecticide and the presence of the prey had little impact on predator mortality levels, the authors concluded that the mortality resulted from an ingestion of the poisoned sap by the predator species.

Soil flora are the only group of terrestrial organisms which are not adversely affected by the use of carbofuran. Hubbell et al. (1973) studied the toxicity of pesticide combinations in the field and found that normal application rates of carbofuran brought about only a marginal reduction in bacterial and fungal populations. Higher than normal application rates caused a 25 percent reduction in nitrification during the eight weeks which followed application. In other studies carbofuran has proved to be actually beneficial to soil microorganisms, with normal application rates resulting in 100 to 300 percent increases in bacterial and actinomycetal populations (Mather et al. 1976).

Carbofuran: Persistence--

Carbofuran is a relatively nonpersistent insecticide (Table 9-6) whose degradation occurs through both biological and non-biological pathways. Hydrolysis, hydroxylation, oxidation, and conjugation metabolic pathways all contribute to the degradation of carbofuran in plants, insects, and other animals (Getzin, 1973). Persistence studies on carbofuran in corn have shown residue levels to be below established tolerances both at the silage stage and at harvest (Caro et al. 1973).

Soil degradation rates of four months (Table 9-6) have given carbofuran the label of a moderately persistent pesticide although its degradation depends greatly on such factors as manner of application, soil pH, and microbial populations. Caro et al. (1973) found persistence to be higher when carbofuran was applied in the furrow to an acidic, heavy textured soil. Half-life in this case was 117 days vs. the 46-day half-life of the material when it was broadcast on a more neutral soil. Getzin (1973) showed a similar relationship of persistence to soil pH and found that carbofuran's half-life extended to 54 weeks in organic muck soils. Getzin also pointed out the importance of microbial degradation of carbofuran; Williams et al. (1976) noted that high microbial activity can be so great as to reduce the insecticidal effectiveness of the compound. Work by Read (1969) indicates that carbofuran may be more persistent than is commonly assumed, band applications retained all but six percent of their larval toxicity 90 days after treatment. Williams and Brown (1976) have found similar evidence of slow degradation in British Columbia with a build-up of residues occurring after two successive years of treatment.

Studies in model ecosystems have shown that carbofuran hydrolyzes rapidly in aquatic systems (Yu et al. 1974) and degrades to water soluble materials which do not persist or remain at high levels (Sanborn, 1974). Applications to flooded rice fields showed maximum concentrations 14 hours after application whereupon residues quickly dissipated, having a half-life of one day or less (USEPA, 1976).

Carbofuran: The Effectiveness of SWCPs in Reducing Transport--

Although very few studies have been done on the effects of SWCPs on carbofuran's runoff, one can estimate the effects of their implementation by looking at the data on other pesticides. Atrazine, for instance, has been used in numerous studies on the effects of SWCPs and shares several physical and chemical characteristics with carbofuran. Although atrazine persists for longer periods of time in the soil (16 to 48 weeks vs. 16 weeks for carbofuran), carbofuran is more soluble (700 ppm vs. 70 ppm for atrazine) and thus is transported more readily in runoff. Studies on atrazine's runoff from agricultural fields report losses that generally fall within the range of 1 to 5 percent of the applied material; Caro et al. (1973) reported annual losses of 0.5 to 2.0 percent of the carbofuran applied to corn grown on silty loam soils, and Smith et al. (1974) reported single runoff event losses of up to 14 percent of the carbofuran applied to corn grown on claypan soils. If the data on the use of SWCPs to reduce atrazine runoff are used, it will be predicted that SWCP could indeed help to reduce the runoff losses of carbofuran (see Tables 9-9 and 9-10 in atrazine section). Stripcropping, ridged corn, and conservation tillage systems brought about reductions of two percent to 79 percent in atrazine's runoff. The modelling studies of Beyerlein and Donigian (Appendix F) indicate that no tillage reduces atrazine losses by twelve percent, contours by twenty-four percent, and terraces with contours by 64 percent. Reductions in carbofuran's runoff losses with no tillage, contours, and terraces with contours may be smaller than the reported reductions for atrazine simply because carbofuran is almost entirely transported in the dissolved phase of runoff. Beyerlein and Donigian reported reductions of six percent, 11 percent and 36 percent in total runoff water with no tillage, contours, and terraces with contours, respectively; reductions in carbofuran's runoff losses would probably be between the reported reductions in atrazine runoff and water runoff. However, there may be cases where SWCPs do not aid in the reduction of carbofuran's runoff losses. It has been shown by Smith et al. (1974) that on claypan soils the runoff losses of atrazine and carbofuran from corn grown under conventional tillage will be equal to, or smaller than the runoff losses from corn grown under conservation and no tillage production systems.

Thus it appears that on claypan soils, carbofuran's runoff losses are large enough to warrant the implementation of runoff control measures; however in the case of claypan soils, the soil and water conservation practices of conservation tillage and no-tillage have no effect or a negative effect on the runoff losses of carbofuran. The impacts of other SWCPs on carbofuran's runoff losses from claypan soils have not been studied. Runoff losses of carbofuran from silty loam soils are generally not large enough to warrant the implementation of control measures although in this case SWCPs could potentially bring about substantial reductions in runoff losses.

Carbofuran: Increasing Efficiency in Use--

Increasing the efficiency of use is not as easily accomplished in the case of carbofuran as it is in the cases of toxaphene and methyl parathion. Because carbofuran is used primarily as a granular formulation for the control of the corn rootworm, drift losses are small and will not usually contribute to the pollution of non-target areas. Simple changes in application procedures thus will not reduce pollution due to carbofuran's use in agriculture.

The cultural control method of crop rotation is an effective means of reducing the need for soil insecticides like carbofuran but is limited in its applicability. Corn rootworms require the continuous presence of corn host plants for the completion of their life cycles, thus annual rotation of corn with another crop eliminates the corn rootworm and the need for soil insecticide treatments for its control. Rotations involving more than one year of corn will not provide complete protection from corn rootworms in years following the first year, although populations are greatly reduced when compared to rootworm populations found in continuous corn. Unfortunately, rotations often mean reduced profits for farmers and thus are not used as widely as might be expected. Corn-soybean rotations are profitable in much of the Midwest; however in this case, the increased sediment and water losses associated with corn-soybean rotations (Section 4) create a conflict between the goals of plant protection and the goals of soil and water conservation.

There is strong evidence indicating that the number of treatments used for the control of corn rootworm could be substantially reduced even without the use of rotations. Having observed that many corn farmers accept soil insecticides as a necessary production input, Turpin and Thieme (1977) monitored rootworm populations in 234 untreated Indiana cornfields and found corn rootworm larvae present in only 34 percent of the test fields. Rootworm populations only reached economic injury levels in ten percent of the infested fields, thus the authors concluded that "prophylactic use of soil insecticides on corn is seldom profitable in Indiana." Integrated pest management programs are now being developed and implemented to help farmers determine when it is necessary to use insecticides like carbofuran for the control of corn rootworm; however, these IPM programs are not as well developed or as accessible as they in the case of cotton. Pilot IPM programs involving scouting and a corn variety resistant to European corn borer are being implemented with much success throughout much of the corn belt. Luckmann (personal communication, 1979) studied these IPM programs in Illinois and found that fewer than 50 percent of the scouted fields had rootworm populations that required insecticide treatments. In former years without scouting, all corn fields would have received an insecticide treatment for corn rootworm whether or not injurious populations were present. Since insecticide treatments cost more than twice as much as scouting, it appears from Luckmann's studies that reductions of up to 50 percent in the use of carbofuran can be gained at no net cost to the farmer.

Carbofuran: Summary and Conclusions--

In contrast to the other two insecticides studied, most of the transport of carbofuran away from target areas occurs with runoff. The major transport routes of these insecticides differ primarily because of differences in formulation and application procedures. Instead of being sprayed by aerial equipment as methyl parathion and toxaphene are, carbofuran is applied primarily in a granular formulation by ground equipment and consequently suffers only small losses through drift. The amount of carbofuran leaving agricultural fields in runoff generally consists of less than two percent of the material applied; methyl parathion and toxaphene have annual runoff losses of 0.5 percent or less of the applied material. However, because carbofuran is applied only once a year and the other insecticides are applied as much as twelve times in one growing season, the annual runoff losses of carbofuran are relatively small and generally consist of less than 30 g/ha. The effects of these concentrations on aquatic organisms are severe but relatively ephemeral as carbofuran has a half-life of only one day in water. Thus, given the small amounts of carbofuran transported in runoff and the rapid degradation of the material in water, carbofuran does not appear to be as hazardous as toxaphene to aquatic ecosystems.

Strategies for reducing pollution are somewhat different in the case of carbofuran simply because the air-borne losses of carbofuran cannot be substantially reduced as they can in the cases of toxaphene and methyl parathion. Any residues of carbofuran found in surface waters will probably have resulted from runoff losses or improper disposal of tank rinsate or carbofuran containers; thus reductions in carbofuran pollution will only come about as a result of programs which improve disposal techniques, reduce runoff losses or decrease the amount of carbofuran applied. Because almost all carbofuran in runoff is in the dissolved phase, the only SWCPs which will significantly reduce carbofuran's transport are those which reduce the amount of runoff water leaving target areas. Although very few studies have been done on the impacts of SWCPs on carbofuran's runoff losses, data on other pesticides sharing solubility and persistence characteristics with carbofuran indicate that terraces with contours could reduce runoff losses by 60 percent. As discussed in Section 8, terraces with contours vary in costs throughout the country but are quite expensive in most cases. Less expensive SWCPs such as conservation or minimum tillage would be expected to reduce runoff losses of carbofuran to a much lesser extent. Simple exchanges in application procedures may prove to be an easier method of reducing the runoff losses of carbofuran; one study on runoff has indicated that runoff losses will be smaller for a given volume of runoff when band (in-furrow) applications instead of broadcast applications of carbofuran are made. Further studies should be conducted to establish the impact and cost effectiveness of such changes in application procedures.

Another strategy for reducing carbofuran losses involves the implementation of pest management techniques which reduce the need for carbofuran applications. For example, the use of carbofuran for corn rootworm control (its primary use) can be completely eliminated if corn is rotated with a crop which does not serve as a host for corn rootworm larvae. In most of the major corn and soybean producing regions, the corn-soybean rotation produces substantial economic returns while eliminating the need for soil insecticide treatments for the rootworm. Unfortunately, sediment losses from soybean fields are quite high; thus if such a rotation scheme resulted in higher acreages of soybeans, it would also contribute to increased sediment pollution. On farms where both corn and soybeans are grown, rotating corn and soybean fields without increasing soybean acreages would not necessarily increase erosion. Sod-based rotations reduce sediment pollution as well as the need for soil insecticides like carbofuran but are not as profitable as continous corn in most areas. Another IPM alternative is the use of scouting programs to monitor pest population levels. This IPM method shows considerable promise for reducing insecticide application rates and frequency of application. In pilot programs on limited acreages in Indiana and Illinois, applications of soil insecticide were reduced by over 50 percent when fields were scouted to determine whether densities of corn rootworm were large enough to warrant an insecticide treatment. With such a reduction in application rates, the cost of scouting is more than offset by the savings in insecticide costs. It should be noted that scouting does not reduce insect densities but does help determine whether insecticide treatment is necessary. Hence in those areas where climate and cropping practices are such that high population levels of corn rootworm occur every year, scouting is not likely to significantly reduce insecticide use. Entomologists working in corn growing areas have estimated that such areas comprise only a small fraction of the total corn acreage currently treated with insecticides for the control of corn rootworm.

In conclusion, both SWCPs and integrated pest management programs can significantly reduce carbofuran losses in runoff. Both strategies also have additional benefits:

1) SWCPs may reduce sediment and nutrient pollution while conserving valuable topsoil and water;

2) the implementation of IPM programs and the resulting decrease in the use of carbofuran would be expected to also reduce the potential for pollution due to improper disposal and for accidents involving farm workers and non-target species living within and outside of target areas. Crop production costs and energy use may also be reduced if IPM programs lead to reductions in pesticide use.

Detailed information on the effectiveness or costs of SWCPs and IPM is not available; however in most areas of carbofuran's use, it appears that IPM is more cost-effective than any of the SWCPs in reducing carbofuran's transport away from agricultural areas. In the remaining areas, the implementation of SWCPs for the control of carbofuran's runoff losses should be preceeded by a careful analysis of the environmental hazards associated with the small and ephemeral concentrations of carbofuran transported to surface waters with runoff.

Atrazine

Atrazine, a s-triazine herbicide, accounts for almost 50 percent of all herbicides used in corn production and is, as stated previously, the most heavily used herbicide in the United States (Adrilenas, 1974). Its high degree of selectivity allows for applications to the soil both before and after the emergence of the crop and greatly reduces the need for time-consuming tillage and cultivation operations.

Atrazine: Runoff and Leaching--

The quantity and concentration of atrazine found in runoff depends on many factors among which are: intensity of rainfall (Baker and Johnson, 1977; White et al. 1967), site characteristics (Bailey et al. 1974), land management and conservation practices (Hall 1974; Ritter et al. 1974; Smith et al. 1974; Baker and Johnson, 1977; Triplett et al. 1978), application rates (Hall et al. 1972) and timing of first runoff producing storm relative to application (White et al. 1967; Hall 1974; Ritter et al. 1974; Smith et al. 1974). This last factor is probably the most important determinant of the actual quantitites of atrazine that will leave a field via runoff. Baker and Johnson (1977) found that seasonal losses were less than five percent in years when the first runoff producing storms occurred two weeks or more after application. However, in one year in which a storm took place 24 hours after application, the losses were 16 percent of the applied atrazine.

The runoff of atrazine from agricultural lands has been studied intensively under a variety of geographic and experimental conditions. von Rumker et al. (1975) have reviewed these numerous studies and have estimated that of the 24 million kilograms of atrazine applied to the 1971 U.S. corn crop, 165,000 to 945,000 kilograms may have been lost in the dissolved and solid phase of runoff. The data from several field and modelling runoff studies are summarized in Table 9-9. These studies vary in terms of the reported

TABLE 9-9. ATRAZINE LOSSES IN RUNOFF

Amount Applied (kg AI/ha)	Type Application	Average Slope (%)	Loss in Runoff (% Application)	Concentration in Runoff Water (ppm)
3.36[1]	surface with simulated rainfall[a]	6.5	2 (2 - 7.3)	0.16-8.08
2.2[2]	surface[b]	14	2.5	0.0-0.8
1.68[1]	incorporated with simulated rainfall[c]	2.2	6.44	0.0-3.3
1.68[1]		3.6	12.47	0.0-7.9
3.36[1]		2.5	13.3	0.0-11.1
3.36[1]		5.7	10.18	0.0-4.0
2.2[2]	surface[d]	14	5.0	0.0-2.3
4.5[2]			4.8	0.05-4.6
3.36[3]	surface[e]	10-15	2.7-16.0	1.17-4.91
2.24[2]	surface[f]	12-18	>5 (>5-15)	---
1.12-3.36[2]	incorporated[g]	8-22	0.02-5.7	0.10-0.48

[1] Duration of experiment was 1 hour on fallow lands

[2] Duration of experiment was 5 months on corn crop.

[3] Duration of experiment was 58 days on corn crop.

[a] White *et al*. (1967).

[b] Hall *et al*. (1972).

[c] Bailey *et al*. (1974).

[d] Hall (1974).

[e] Ritter *et al*. (1974).

[f] Baker and Johnson (1977).

[g] Triplett *et al*. (1978).

quantities and concentrations of atrazine found in runoff, but all note that atrazine concentrations are much higher in the sediment portion of the runoff. Baker and Johnson (1977) found atrazine concentrations in sediment runoff were five times as large as those in the water runoff, but pointed out that in terms of total quantity lost, more atrazine is associated with the water portion as water losses are much larger than sediment losses. Baker and Johnson go on to say that because the major herbicide loss occurs with water, erosion control practices will not prevent the bulk of atrazine runoff losses.

Researchers have differed in their evaluations of the ecological significance of atrazine's runoff losses. Some have conjectured that runoff waters from non-treated areas and surface waters will dilute atrazine-laden waters and reduce hazardous herbicide concentrations to levels below acute toxicity (White *et al.* 1967; Hall, 1974; Triplett *et al.* 1978). This argument ignores the possibilities of sublethal effects on non-target species and disruptions in food webs caused by reduced densities of algal and microbial populations. Such possible sublethal and secondary effects should be further investigated for this ignorance, coupled with the lack of data on atrazine's degradation in water, makes it difficult to evaluate the environmental impact of atrazine's runoff losses.

The rate at which atrazine leaches through agricultural soils varies greatly according to the following factors: plant adsorption (Hall and Hartwig, 1978), reversible and irreversible adsorption (Bailey and White, 1970; Sanborn *et al.* 1977; Hall and Hartwig, 1978), degradation rates (Hall and Hartwig, 1978; Harris *et al.* 1969), soil composition and moisture (Burnside *et al.* 1971; Bailey and White, 1970; Sanborn *et al.* 1977); cropping practices, and nutrient levels (von Stryk and Bolton, 1977). Leaching losses would be expected to be greatest from light soils with small adsorption and/or degradation capacities (Sanborn *et al.* 1977).

Different combinations of the above factors have yielded data showing a considerable range of leaching depths for atrazine residues. Burnside *et al.* (1963, 1971) have found atrazine residues well below the plow layer, and have cited leaching as a major factor in the removal of the herbicide from agricultural areas. The possible hazards from atrazine leaching losses are well illustrated by Leh's work (1968) in which groundwater was found to be polluted by atrazine in wet years and during periods of application. Richard *et al.* (1975) also detected atrazine in all of the wells sampled in the alluvial plains of Iowa rivers contaminated with atrazine. Other studies have shown that the bulk of applied atrazine reacts and dissipates in the plow layer with only small amounts reaching lower levels in the soil profile (Hall and Hartwig, 1978; Baker and Johnson, 1977; Schwab *et al.* 1973; von Stryk and Bolton, 1977). von Stryk and Bolton (1977) studied the leaching of atrazine into tile drains under rotation corn and continuous corn with higher fertility levels and found that although losses were higher in the continuous corn, both cropping systems had relatively small losses of 0.25 percent and 0.8 percent, respectively.

Atrazine: Air-borne Losses--

The magnitude of atrazine's drift losses depends on weather conditions and application procedures (Table 9-8). von Rumker et al. (1975) have reviewed the available literature on drift losses of herbicides and have constructed general drift loss tables for any herbicide based on the application procedure. According to their table, atrazine losses through drift could range from negligible for ground equipment applications, to 40 percent for application by air equipment. A field study by Baker and Johnson (1977) shows that significant losses of atrazine can occur using ground equipment under windy conditions. Immediately after application measurements were made of the amounts of herbicide deposited on the soil and on filter paper. Results showed that with winds of 20 km/hr during application, losses of atrazine range from 48 to 51 percent of the applied material. Unfortunately, it is not known how often farmers apply pesticides under such windy conditions or how large drift losses are in wind speeds more typical for spraying operations.

Volatilization losses can also be quite high when atrazine is surface applied to warm, dry soil, and is not incorporated via cultivation, rainfall or irrigation. Kearney et al. (1964) measured losses under such conditions and reported that with temperatures of 35°C, losses were as high as 40 percent during a 72 hour period. With a temperature increase to 45°C, they reported losses of nearly 80 percent of the material originally applied. Foy's (1963) experiments pointed to a similar relationship between volatility and temperature, with temperatures of 40°C and 60°C bringing losses of 11 percent and 95 percent, respectively.

The environmental fate and significance of atrazine's air-borne losses are not well understood. Atrazine drift reaching non-target plants could result in damage similar to the decline of the hackberry seen in towns receiving herbicide drift from corn growing regions in Iowa (Hibbs, 1976). Other air-borne losses from drift, volatilization, and wind erosion will be redeposited on the earth's surface with rainfall and the settling out of dust particles. Atrazine residues have been found to be associated with suspended solids in rainfall at concentrations of 0.1 ppb (Cohen and Pinkerton, 1966). Further studies establishing the persistence and concentrations of atrazine in rainfall would be extremely beneficial for future evaluations of the importance of controlling air-borne losses.

Atrazine: Toxicity--

Knowing the mode of action of a herbicide helps one to predict which non-target species will be adversely affected by its use. Atrazine, which kills weeds by inhibiting photosynthesis, would be expected to have deleterious effects on phototrophic organisms living in surface waters and the soil. These phototrophic organisms are often important links in food webs, thus their demise may cause reductions in the populations of organisms (such as fish) dependent on them for a food source. A disruption of both plant and animal populations may also occur as a result of the habitat changes brought about by atrazine use.

Generally, those conditions which allow for the effective herbicidal action of atrazine are identical to those conditions which make for the greatest inhibition of soil algal populations. Atrazine-treated soils rich in organic matter apparently absorb and inactivate larger amounts of the herbicide, and thus escape a severe inhibition of their algal populations (Kaiser *et al*. 1970). Atrazine is also removed from the sphere of biological interactions in highly acidic soils. The hydrogen ions prevalent in these soils associate with the herbicide to form positively charged substances that are readily adsorbed by soil particles (Weber *et al*. 1975). Kaiser *et al*. (1970) have found that while small dosages of atrazine may actually stimulate microflora, large applications inhibit growth. They go on to predict that the decreases in soil organic matter and weed populations brought about by herbicide use will further inhibit soil microbial populations.

Atrazine has also been shown to affect arthropod population levels. Springtails, mayflies, caddisflies and leeches have all been adversely affected by low levels of atrazine (Thompson, 1973). Lichtenstein *et al*. (1973) have shown that atrazine can have a synergistic effect on the toxicity of insecticides. In experiments with parathion, the addition of normally non-toxic atrazine increased insect mortality from eight to 50 percent. Such synergistic effects must be taken into consideration when evaluating the toxicity of various insecticides to insect predators and parasites.

Atrazine is less toxic to higher organisms. It is relatively non-toxic to birds and mammals (Table 9-5), the latter having efficient excretion and degradation systems for the herbicide (von Rumker, *et al*. 1974; Esser *et al*, 1975). Atrazine is considered to be moderately toxic to fish (Table 9-5) with lethal concentrations for 50 percent of the population ranging from 0.55 - 12.6 ppm, levels quite within the range of concentrations seen in surface runoff. Although little work has been done on the more insidious sublethal effects on terrestrial and aquatic organisms, the aforementioned work by Plewa and Gentile (1976) showing atrazine to be mutagenic to yeast does not allow for complacency. Yoder *et al*. (1973) have further shown that custom applicators of herbicides (including atrazine) have up to four times as many chromosome aberrations in their blood lymphocyte cultures as compared to a control population.

Haque *et al*. (1977) have used a bioaccumulation ratio to evaluate the biomagnification of atrazine in aquatic food chains. This ratio compares the concentration of the chemical in the organism to the concentration of the chemical in the water. Values obtained for atrazine were two to 15 for snails, ten to 83 for algae and three to ten for fish. Metcalf *et al*. (1971) conducted biomagnification studies in a terrestrial-aquatic model ecosystem and concluded that magnification is unlikely to occur in aquatic food chains.

Atrazine: Persistence--

Atrazine is generally applied to the soil surface, although incorporation increases its activity and prolongs its effectiveness. Once in the soil, atrazine is reversibly adsorbed to soil particles, with desorption occurring readily, although at a much slower rate than that of adsorption. This steady desorption releases atrazine into the soil solution and allows excellent continuing control of weeds but also creates residue problems in the soil environment. At times, atrazine residues have accumulated in high concentrations and have caused damage to susceptible crops planted in following years. In many other crop situations, factors such as soil temperature (Haque and Freed, 1974), pH, moisture (Weber and Weed, 1974), crop cover (Hiltbold, 1974), rate of application (Libik and Romanowski, 1976) and herbicide depth in the soil (Harris *et al.* 1969) interact to create conditions that are more optimal for degradation of atrazine. In any case, atrazine has proven to be a fairly stable substance in the soil with persistence rates of four months to one year (Haque and Freed, 1974). In the 1971 National Soils Monitoring Program, Carey *et al.* (1974) found that, of the 213 soil samples taken in areas treated with atrazine during the growing season, 71.4 percent contained residues of the material. LeBaron (1970) has suggested that atrazine residue problems could be reduced by plowing and thorough mixing of the soil following crop harvest. LeBaron also indicates that soils could be detoxified with the use of charcoal but adds that this technique is not yet feasible because of unresolved economic and application problems.

Surprisingly enough, an extensive review of the literature uncovered little information on the persistence of atrazine in water although studies showed that degradation does take place via photodecomposition and hydroxylation (Jordan *et al.* 1970). Khan (1978) has shown that the chemical hydrolysis of atrazine in solution is pH dependent, proceeding much slower at those pHs associated with soil solutions and surface waters and more rapidly in waters originating from coniferous forests. Studies of atrazine and its hydrolysis product, hydroxyatrazine, under conditions of limited aeration have shown that the total breakdown of atrazine will be slower in submerged sediments and faster in aerated soils (Goswami and Green, 1971).

Data have been published which establishes the presence, be it persistent or transient, of atrazine in aquatic systems. In a study by Morley (1977), atrazine was found in four out of six watersheds, with mean concentrations peaking at 5 ppm in June and declining to 2 ppb in July and 0.5 ppb for the reminder of the year. In another study by Richard *et al.* (1975) samples were taken in Iowa to ascertain the extent and the degree of contamination of surface, sub-surface and finished drinking waters by dissolved DDE, dieldrin, and atrazine. Of the three pesticides monitored, atrazine appeared in the highest concentrations, and was found in several shallow wells and in water which had been ostensibly decontaminated by filtration through activated carbon beds. The contaminated shallow wells were all located in the alluvial plains of contaminated rivers. It is difficult to determine the importance of these residues without degradation data, for if degradation is fairly rapid, the trasitory presence of the herbicide may not warrent extensive pollution control plans to prevent it from entering surface waters.

Atrazine The Effectiveness of SWCPs in Reducing Transport--

Several studies have examined the impact of SWCPs on atrazine's runoff losses from agricultural fields (Table 9-10). In the majority of cases the impact of a SWCP was significant; however, in areas with claypan soils the amounts of atrazine measured in runoff from fields with conservation tillage exceeded the atrazine levels measured in runoff from fields with conservation tillage (Smith *et al.* 1974). Stripcropping provides the most effective control and reduced atrazine's runoff losses by 79 percent (Hall, 1974). To incorporate the effects of changing weather conditions, a simulation model of atrazine transport was developed by

TABLE 9-10. EFFECT OF SWCPs ON ATRAZINE LOSSES IN RUNOFF

Tillage Systems[2] Compared	Application[1] Rate (AI/ha)	Runoff Losses Under Conventional Tillage (g/ha and % of applied)	Runoff Losses Under Conservation Practice (g/ha)
conventional I[a]	2.2 kg/ha[3]	77 g/ha = 3.5%	10 g/ha
vs.			
stripcropping	4.5 kg/ha	49 g/ha = 1.1%	17 g/ha
surface contoured[b]	3.36 kg/ha[4]	275 g/ha = 8.2%	131 g/ha
vs.			
ridged corn		537 g/ha = 16.0%	91 g/ha
conventional II[c]	3.25 kg/ha[5]	17.3 g/ha = .53%	32.5 g/ha
vs.		19.7 g/ha = .61%	76.8 g/ha
no-till 1		41.6 g/ha = 1.28%	52.3 g/ha
		80.6 g/ha = 2.48%	67.9 g/ha
conventional III[d]	2.24 kg/ha[3]	20 g/ha = .89%	18 g/ha
vs.			
till plant			
conventional III[d]	2.24 kg/ha[3]	20 g/ha = .89%	40 g/ha
vs.			
ridge plant			
conventional IV[e]	1.12 kg/ha[3]	0.35 g/ha = .03%	0.27 g/ha
vs.			
no-till 2			
conventional V[f]	2.91 kg/ha[6]	53.51 g/ha = 1.8%	47.24 g/ha
vs.			
minimum tillage	2.80 kg/ha[6]	79.24 g/ha = 2.8%	76.81 g/ha

TABLE 9-10. (Continued)

Tillage Systems[2] Compared	Application[1] Rate (AI/ha)	Runoff Losses Under Conventional Tillage (g/ha and % of applied)	Runoff Losses Under Conservation Practice (g/ha)
conventional V[f]	2.91 kg/ha[6]	53.51 g/ha = 1.8%	40.88 g/ha
vs.			
contour tillage	2.80 kg/ha[6]	79.24 g/ha = 2.8%	62.97 g/ha
conventional V[f]	2.91 kg/ha[6]	53.51 g/ha = 1.8%	18.96 g/ha
vs.			
terraces with contours	2.80 kg/ha[6]	79.24 g/ha = 2.8%	29.76 g/ha

[1] Applied as 80% wettable powder.

[2] Descriptions of tillage systems:

Conventional I = rototilling to 15 cm and soil smoothed by hand

Stripcropping = inclusion of oatstrip adjacent to corn at slope base.

Conventional II = land prepared by turning with a moldboard plow, discing, harrowing plus cultivation of row crops.

No-till 1 = crop residues from previous years are chopped and remain on the surface. Seed is planted through these residues with no additional land preparation. Weeds were controlled by use of herbicides.

Conventional III = spring plowing, discing and surface planting. 3% residue cover.

Till Plant = buffalo till planting in which previous years' ridges were split at plow planting time. 20% residue cover.

Ridge Plant = technique which retained the same range from year to year. 45% residue cover.

FOOTNOTES FOR TABLE 9-10. (Continued)

Conventional IV = plowed to depth 20 cm, disced and harrowed. One post planting tillage of rotary hoeing or cultivation was used.

No-till 2 = corn planted in residues from previous crop with planter that disturbed band of soil 7-10 cm wide. Weeds controlled by herbicides.

Conventional IV = disced and planted in straight rows parallel to the slope. Crop removal in the fall.

Minimum Tillage = no till prior to spring planting. Planting done with crop residue remaining from fall, rows parallel to slope.

Contour tillage = disc in spring prior to planting, rows perpendicular to slope. Residues removed after harvest.

Terrace with Contours = disc in spring prior to planting, watershed area of 1.29 hectares is divided into 2 terraces with contoured rows perpendicular to slope.

[3] 5-month field study on corn.

[4] 58-day field study on corn.

[5] 3-month field study on corn.

[6] 10-year simulation study on corn.

[a] Hall (1974).

[b] Ritter et al. (1974).

[c] Smith et al. (1974).

[d] Baker and Johnson (1977).

[e] Triplett et al. (1978).

[f] Appendix F.

Hydrocomp, Inc. (Appendix F). The model was computed over a long period of time and a range of weather conditions. The atrazine losses estimated by the simulation model indicate that less than 1.84 percent of the material applied is carried by runoff from a field with conventional tillage. The use of SWCPs reduced this loss to 0.65 percent using terraces with contours. Less expensive SWCPs such as no tillage and contour tillage bring about smaller reductions in atrazine's runoff losses.

Atrazine: Improving Efficiency in Use--

Finding methods to increase efficiency in the use of herbicides is in many ways a more difficult task than it is with insecticides. High costs and crop injury resulting from the overuse of herbicides have worked together in a fortuitous manner to prevent farmers from the sometimes wasteful use of pesticides seen in the case of cotton insect control programs. However, although weed control has not constituted a large portion of IPM research and implementation, there is some indication that improvements could be made in the use of herbicides. The careful monitoring of weed population sizes, growth stage, and species type would help farmers make decisions concerning the type and rate of herbicide needed and whether treatment is necessary. Research establishing what size weed population can be tolerated in a field without significant damage to crops might persuade farmers to abandon the "clean" field as a goal in weed control.

Research to determine when the weed species is most susceptible to chemical treatment could also potentially reduce the amounts of herbicides needed for weed control. Recent work has shown that weather conditions can influence the effectiveness of the triazine herbicide metribuzin in that the tolerance of both tomatoes and jimsonweed declines following cloudy days. Fortunately, the decline in tolerance is greater in the jimsonweed. In this case, a reduction of 40 percent in the application rate of metribuzin still provided effective weed control after three days of cloudy weather (Seim, 1978). Crosby (1975) has noted that the herbicidal powers of atrazine are similarly inactivated in sunlight. The development of application equipment with improved placement and conservation of herbicidal materials will further decrease the amounts of pesticides available for transport to surface waters (McWhorter, 1977; Seim, 1978).

It is only quite recently that attempts have been made to investigate the use of integrated pest control methods to suppress weed populations in field crops. The success of biological control of Klamath weed by an insect herbivore indicates that there is a potential for integrated pest managment for control of agricultural weeds as well. However, at the present time it is not possible to predict how effective such programs will be or when they can be implemented.

Unfortunately, the use of atrazine can interfere with integrated pest management programs for insect control. Harris and Miles (1975) have noted that the use of persistent herbicides such as atrazine encourages the farmer to grow corn year after year because other crops such as soybeans may be damaged by residues remaining in the soil the following year. Unfortunately, the corn rootworm thrives under such continuous corn programs; increases in its population levels have necessitated the widespread use of insecticides for rootworn control. The cultural control method of crop rotation, which formerly controlled the rootworn, thus cannot be utilized in agricultural areas producing rotation crops susceptible to damage by atrazine residues.

Atrazine: Summary and Conclusions--

The effectiveness of SWCPs for reducing pesticide transport in runoff has been studied more extensively for atrazine than it has been for any of the other four representative pesticides. The results of these studies indicate that atrazine losses in runoff with conventional tillage are generally rather small (one to five percent) and that SWCPs can reduce these losses. Stripcropping and terracing with contours appear to be the most effective SWCPs for this purpose (Table 9-10). However, it is possible that the use of a SWCP to decrease the amount of atrazine in surface runoff will be accompanied by an increase in atrazine's leaching losses. Modelling studies have predicted that atrazine will not leach into groundwater, but in empirical studies atrazine has been detected in groundwater. Thus, it is not clear that SWCPs would actually decrease water contamination by atrazine in all areas.

Since pest control techniques requireing less atrazine use are not currently available, the only other pollution control strategies beside SWCPs are control of drift and volatilization or the replacement of chemical weed control with cultivation. The studies measuring atrazine losses from volatilization and drift reported losses ranging from negligible to greater than 40 percent. The procedures for reducing air-borne losses include incorporation of atrazine into the soil and elimination of aerial applications.

As with the other pesticides, the question exists whether air-borne losses are as environmentally hazardous as runoff. The air-borne losses of atrazine are smaller than they are in the cases of methyl parathion and toxaphene primarily because weed treatments are generally made with high volume large droplet applications by ground equipment. Unfortunately, there were not enough data over a range of wind and temperature conditions to determine the exact magnitude of air-borne losses; however, available evidence suggest that drift and volatilization losses combined probably exceed the one to five percent of the material commonly transported by runoff. The environmental impact of air-borne and water-borne atrazine also depend upon atrazine's persistence in water and air. Although several studies have shown that atrazine is quite persistent in soil, little information is available on its persistence in water or air.

In conclusion, it appears that SWCPs are among the most effective means of reducing potentially hazardous concentrations of atrazine in runoff. However, before wide scale implementation of SWCPs occurs, studies on aerial inputs of atrazine to terrestrial and aquatic ecosystems should be done to establish whether it would be more cost effective to reduce environmental polution from atrazine use by reducing drift and volatilization losses. Simple changes in spraying equipment or changes in atrazine's placement in the soil may also prove to be more cost effective means of reducing atrazine's runoff losses. Furthermore, empirical studies on a range of soil types should determine whether SWCPs which increase percolation will increase the leaching losses of atrazine.

Paraquat

Paraquat is a contact bipyridylium herbicide whose chemical properties and non-selective mode of action make it highly effective as a desiccant used in the harvest of cotton, potato haulm, and sugar cane, and as a herbicide for grass and broadleaved weed species in both aquatic and agricultural ecosystems. Rapid action and a non-residual nature make it well adapted for use in no tillage and conservation tillage corn production systems, where it has been used extensively to destroy weeds before the planting of the crop or before crop emergence.

Paraquat: Runoff and Leaching--

Although substantial amounts of paraquat may leave agricultural fields with runoff, the physical and chemical characteristics of the herbicide greatly reduce the hazards commonly associated with these losses. When paraquat is applied to a field, a portion of the material decomposes or is adsorbed into plant tissues. The paraquat which reaches the soil is irreversibly adsorbed to clay particles and reversibly adsorbed to organic matter. In the weeks following application, the paraquat initially adsorbed to organic matter is redistributed to the clay particles, which are believed to be the final repository of paraquat residues. If a runoff producing storm occurs during this period of redistribution, unattached paraquat could potentially leave fields in runoff water; however, this is apparently not the case for paraquat is rarely found in runoff water.

Researchers conducting field and modelling studies on paraquat's runoff have reported both large and small losses from agricultural fields. Smith *et al.* (1978) found solid phase runoff losses which comprised 22 percent of the applied material; however, because paraquat is only applied to foliage when used in agriculture and the authors applied the material directly the soil, Smith *et al.* stated that these losses do not represent the range of losses expected *with* normal agricultural [illegible]. Studies in Iowa have found smaller losses of 1.28 percent of the applied material (Baker *et al.* 1979). Modelling studies based on data from the Georgia watershed *studied* by Smith *et al.* (1978) predicted losses of 340.6 g/ha of the 2520 g/ha of paraquat *applied* to the soil (Appendix F). As discussed below, there is no indication that soil-bound residues of paraquat will desorb from soil or enter into biological interactions upon reaching aquatic systems (Calderbank and Slade, 1976).

Paraquat: Air-borne Losses--

Although some aerial application of paraquat does occur, fields are generally treated with high volume, large droplet applications with ground equipment which presumably do not produce hazardous amounts of drift. However, recent studies indicate that small but serious drift losses of herbicides can occur even when applications are made by ground equipment in light winds. Byass and Lake (1977) studied drift losses of paraquat resulting from normal ground equipment application procedures used in Great Britain and correlated percent drift losses with damage to crops located downwind of spraying operations. They found that drift losses which comprised more than 0.1 percent of the applied paraquat caused significant damage to bean plants located downwind of the treated area; the downwind area at risk for bean plants was up to one meter in a light wind and up to 20 or 30 meters when paraquat was applied in the highest wind speed in which spraying would normally be attempted. Hazards from paraquat drift may be extended for hundreds of meters if crop or non-target species located downwind are more sensitive than bean plants are to paraquat. Aerial applications of paraquat will result in higher drift losses and serious hazards to non-target areas. Although no studies have been done specifically on paraquat's drift losses with aerial applications, von Rumker _et al._ (1975) constructed general drift loss tables based on application procedures and estimated that ten to 40 percent of the herbicides applied to the U.S. corn crop by aerial equipment will drift over 3040 meters from the target area. Volatilization does not contribute to the air-borne losses of paraquat from agricultural areas (Calderbank and Slade, 1976).

Paraquat: Toxicity--

Hazards to non-target species are greatly reduced because paraquat loses its toxicity when adsorbed to clay particles and retains little toxicity when adsorbed to soil organic matter. Paraquat generally does not have adverse effects on most soil organisms or microbial activities important to soil fertility; however, reductions in springtail populations and residues in earthworms have been reported (Calderbank and Slade, 1976; Sanborn _et al._ 1977; Thompson, 1973). Thompson (1973) has concluded that these toxic effects are not drastic and points out that residues in earthworms are excreted when the worms are placed in untreated soil. Although bound paraquat residues may be innocuous, the material is quite toxic in the period between spraying and adsorption to soil particles. Paraquat is extremely toxic to entomophagous mites and insects and has been considered to be too toxic to allow its use in apple pest management programs (Rock and Yeargan, 1973). The material is relatively non-toxic to birds and moderately toxic to mammals (Table 9-5) but poses hazards for spray applicators because of the irreversible, proliferative changes brought about in lung tissue after inhalation (Vettorazzi, 1977).

The incidence of paraquat contamination of aquatic systems via runoff losses is greatly reduced because paraquat is so strongly adsorbed by eroded soil particles. Even when paraquat is directly applied to surface waters for weed control, paraquat is given but a small chance to adversely affect aquatic animals. The herbicide is rapidly adsorbed by weeds and by the hydrosoil, and may remain in aqueous solution for as little as one half hour (Yeo, 19[illegible]7). Once the paraquat has reached the hydrosoil, desorption would not be expected to occur (Calderbank and Slade, 1976). However, in the brief period between application and adsorption, paraquat will have severe effects on bacteria, freshwater algae, and freshwater protozoa at concentrations of 0.6 ppm (Sanborn _et al._ 1977). Bioconcentration has not been a problem in aquatic ecosystems and levels used for aquatic weed control are well below toxic levels for fish (Calderbank and Slade, 1976).

Paraquat: Persistence--

Paraquat's "persistence" can be perceived in several different ways depending on which general definition of persistence one has in mind. If persistence is taken to mean the length of time required for the disappearance of the original form of the pesticide, then paraquat will be seen as extremely persistent with some researchers asserting that degradation of bound residues does not occur (Fryer *et al*. 1975). Perhaps more appropriate to the bipyridylium herbicides, is the definition of persistence as that period of time wherein the pesticide maintains its insecticidal or herbicidal activity. By this definition, paraquat is seen as a non-persistent pesticide since its phytotoxicity is lost on adsorption to the clay particles so abundant in agricultural soils (Riley *et al*. 1976).

The phytotoxicity of paraquat depends on the strong adsorption capacity of the soil and the type of clay minerals present. Weber and Scott (1966) and Weber *et al*. (1969) found that given comparable application rates, phytotoxicity was greater in soils dominated by vermiculite and kaolinite clays than in soils dominated by montmorillonite. This presumably occurred because montmorillonite has a strong adsorption capacity which completely deactivates paraquat residues, whereas vermiculite and kaolinite cause reductions in paraquat's phytotoxicity but release the herbicide in small amounts. This slow release of paraquat residues results in concentrations which can injure cucumber seedlings grown in model soil systems. Tucker *et al*. (1967) leached soils with concentrated salt solutions and found ratios of irreversibly bound-paraquat to reversibly bound-paraquat of 1:4 in loam soils, 1:27 in sandy soils, and 1:107 in muck soils. Riley *et al*. (1976) have concluded that bound paraquat is not available to living organisms, thus whatever damage is seen is due to free paraquat in the soil solution.

Numerous studies have been done on the abilities of agricultural soils to adsorb and deactivate applied paraquat after continued use. Some researchers have shown that would take 50 to 100 years to saturate the strong adsorption capacity of the top 2.54 cm of the least adsorptive soil considered (Boon, 1964; Riley *et al*. 1976). Riley *et al*. (1976) have calculated that if 100 percent of the active ingredients of a 1.0 kg/ha application reached the soil surface and was uniformly incorporated in the top 15 centimeters of the soil, concentrations of 0.5 μg paraquat/gram soil would result. They also have reported the results of studies done by the Chevron Chemical Company in the United States and the ICI Plant Protection Division in the United Kingdom; these studies indicated that almost all agricultural soils can bind from 50 to 500 μg paraquat per gram of soil and that even the least adsorptive soils (sandy soils and peat soils) can bind at least 50 μg paraquat per gram of soil. Calderbank and Slade (1976) have stated that desorption of these bound residues would not be expected to occur either in agricultural soils or from bottom mud.

Residue problems may occur in soils containing only a small percent of deactivating clays. Juo and Oginni (1978) have pointed out that although farmers in temperate regions have had no phytotoxicity persistence problems with paraquat, soils in the humid tropics contain smaller amounts of deactivating montmorilloinite clays. Damanakis *et al*. (1970) have indicated that residue problems could also occur in organic soils where paraquat can be made available to plants in amounts as low as one twenty-fifth of the strong adsorption capacity. Calderbank and Slade (1976) have reported residual activity of paraquat in organic soils for up to two weeks after application and have suggested that this period indicates the amount of time necessary for the paraquat initially adsorbed on the organic matter to redistribute itself onto the deactivating clay minerals. Weber and Best (1972) have also shown that paraquat is more phytotoxic in soils that have been heavily limed as calcium cations compete with paraquat cations for adsorption sites on solid colloids, leaving more "free" paraquat in the soil solution for plant uptake. Because these phytotoxic effects lasted only several weeks, it was hypothesized that all the paraquat was eventually adsorbed by the soil particles with strong adsorption capacities.

Degradation of paraquat is greatly hindered by its strong adsorption on clay particles; however, a certain amount of the material does degrade on plant surfaces and in the soil solution. Slade (1966) has shown that before its rapid adsorption into plant tissues, some of the applied paraquat photodecomposes on plant surfaces, forming two photolysis products which rapidly degrade soon after formation (Riley et al. 1976). When paraquat reaches soil, microbial and chemical degradation are greatly hindered by the binding of residues to clay particles (Weber and Weed, 1974; Burns and Hayes, 1974; Calderbank and Slade, 1976; Hance, 1967).

Degradation of residues in water is in most cases, only a laboratory phenomenon as any paraquat reaching surface waters is strongly adsorbed by weeds, suspended soil particles, and bottom mud (Way et al. 1971; Juo and Oginni, 1978). Indeed, Coates et al. (1965) have shown that 35 weeks are required for a complete disappearance of paraquat in water, while disappearance from water with sediment and from water with sediment and plants takes six to eight weeks and three to four weeks, respectively. The rapid destruction of plant tissue caused by paraquat does not allow time for metabolic degradation in the plants (Slade, 1966; Funderbank and Lawrence, 1964) thus residues eventually make their way to the bottom mud (Calderbank and Slade, 1976). Photodecomposition of paraquat residues does not occur in aqueous solution (Calderbank and Slade, 1976).

Paraquat: The Effectiveness of SWCPs in Reducing Transport--

Paraquat's strong adsorption to soil particles indicates that SWCPs will be quite effective at reducing paraquat's runoff losses. Field studies measuring the effectiveness of SWCPs at reducing paraquat's transport are not available; however, Beyerlein and Donigian (Appendix F) have used the field data of Smith et al. (1978) to model the effectiveness of no tillage practices, contours, and terraces with contours in reducing paraquat's runoff losses. The model predicted runoff losses with conventional tillage to be fourteen percent of the applied material. Runoff losses with no tillage, contours, and terraces with contours were predicted to be seven percent, nine percent, and six percent, respectively, of the material applied. It should be noted that paraquat's primary use is in providing the additional weed control necessary in no tillage corn. In many cases, paraquat applications can be eliminated by converting to conventional tillage. In such cases, conventional tillage is the most effective means of reducing paraquat losses.

Although SWCPs can substantially reduce the amounts of paraquat transported in runoff, it is questionable whether these runoff losses are hazardous to aquatic systems. As discussed previously, paraquat is carried only on the sediment fraction of runoff and apparently does not desorb from clay particles reaching surface waters. Unless later studies indicate that paraquat runoff losses can be hazardous to aquatic ecosystems, the implementation of SWCPs specifically for the control of paraquat transport does not appear to be a cost effective means of improving water quality.

Paraquat: Increasing Efficiency in Use--

Although the paraquat leaving agricultural fields in runoff does not appear to pose hazards to aquatic ecosystems, losses due to drift or improper disposal of paraquat containers may enter and contaminate surface waters until residues are adsorbed by sediment or aquatic plants. When and if this occurs, increasing the accuracy of placement and improving disposal techniques may prove to be the only pollution control strategies available short of eliminating paraquat use by a return to conventional tillage. Methods for improving disposal techniques have been discussed in an earlier general section on disposal. Strategies for improving the efficiency of paraquat's use in agriculture would be similar to those strategies discussed in the section on atrazine, and would include substitution with ground applications in areas where paraquat is applied by aerial equipment.

Paraquat: Summary and Conclusions--

Before considering the allocation of funds for the implementation of SWCPs for reducing paraquat transport in runoff, the extent and importance of the pollution caused by paraquat in runoff should be determined. On the basis of currently available information, it appears that paraquat in runoff does not cause serious pollution problems. Researchers have reported that paraquat in runoff is completely and irreversibly adsorbed to clay particles and hence is not available to fauna and flora. Thus, although losses of paraquat on eroded soil particles are relatively large, these losses appear to be environmentally innocuous, and little incentive exists for controlling them.

However, there may be unusual situations in which contamination problems will arise. Soils containing high proportions of organic matter and sand adsorb paraquat in a rather weak manner and may exhibit phytotoxicity until the residues are redistributed onto deactivating clay particles. If these soil particles are carried in runoff to aquatic ecosystems, desorption may occur and cause contamination if deactivating clays are not suspended in the water or found in the bottom mud.

The most effective way to reduce paraquat contamination is to eliminate its' use. Since paraquat's primary use is in no till corn production, paraquat's use can be eliminated by changing to an alternative tillage system. Such a change in tillage would not only eliminate the paraquat in runoff but would also eliminate the apparently more serious problems associated with the exposure of farm workers and non-target species to drift losses.

Although only a small fraction of the paraquat used in agriculture is applied by aerial equipment, the drift losses which result from aerial applications appear to pose a more serious hazard to farm workers and to aquatic ecosystems in these areas than do the bound paraquat residues transported in runoff. Hence, in areas where aerial applications are used, their replacement with ground applications would be more effective than any SWCP in reducing paraquat pollution.

With practices such as no tillage which increase application rates, SWPCs are very effective at reducing paraquat losses in runoff. They also do not appear to have any negative impacts such as increasing paraquat's leaching losses. Hence, the implementation of SWCPs for the control of water and sediment losses would not appear to increase paraquat contamination of surface waters. However, for most soil conditions, it does not appear that the hazards posed by paraquat's runoff losses would justify allocation of resources for the implementation of SWCPs specifically for the control of paraquat's transport away from agricultural fields.

RESEARCH NEEDS

The preceding discussions of the five representative pesticides have been based upon currently available information. In some cases this information was not very extensive. Thus, it was often necessary to extrapolate from data on related pesticides in order to draw conclusions about pollution control strategies.

Research in the following areas would be beneficial in evaluations of these five pesticides and would undoubtedly influence decisions regarding the control of their residues in aquatic systems.

1. Methyl Parathion
    a. Additional studies measuring methyl parathion's runoff losses should be initiated to reconcile results from field studies and results from the simulation study done by Hydrocomp (Appendix F).
    b. The environmental and health hazards associated with bound residues of methyl parathion should be evaluated to determine the benefits of reducing the transport of bound residues.
2. Toxaphene
    a. Studies on toxaphene's rate of desorption from runoff-transported sediment would help planners to decide what degree of transport control is necessary.
    b. Studies on the fate, persistence and chemical and physical properties of toxaphene's active moieties would allow monitoring agencies to better evaluate the impact of toxaphene use on terrestrial and aquatic ecosystems.
3. Carbofuran
    a. Field studies should be initiated for a range of soil types to

measure the impact of SWCPs on the amount of carbofuran carried in runoff.

4. Atrazine

   a. If SWCPs are being considered for the control of atrazine runoff, research should be done to determine the impact of increased vegetative cover (associated with minimum tillage techniques) on pesticide degradation rates in soil.

   b. If SWCPs prove to increase levels of pesticide residues in agricultural soils, studies should be done to determine the impact of these residues on soil microorganisms, earthworms, and beneficial arthropod species.

   c. The mutagenicity of atrazine should be further investigated in order to evaluate the hazards associated with the use of the material in agriculture.

5. Paraquat

   a. The spatial distribution of paraquat use should be investigated to determine if there are areas of heavy use where the clay content of the soils is so unusually low that paraquat residues would not be totally adsorbed.

   b. Studies on the ultimate fate of paraquat in soil and in aquatic systems should be carried out to determine if a significant degree of desorption ever occurs.

In addition, research in the following areas would allow environmental quality planners to make more complete evaluations of the hazards associated with pesticide use and would further enable these planners to make more accurate decisions concerning pollution control methods:

1. the amounts of pesticide loss via drift and volatilization from agricultural fields under a range of climatic conditions and application procedures,

2. the rate at which air-borne losses of pesticides are redeposited on the earth's surface,

3. the rate at which pesticide degradation occurs in air and water,

4. the sublethal and secondary effects of pesticides and their toxic synergistic effects with other pollutants,

5. the percent of the reported pollution incidents concerning pesticides which occur as a result of improper disposal techniques, and

6. the effectiveness of changing application procedures to reduce losses of pesticides from agricultural fields.

SUMMARY AND CONCLUSIONS

In most cases SWCPs will reduce the amount of pesticide transported by surface runoff. The exceptions to this rule are those SWCPs which require increases in pesticide use which are larger than the reductions obtained in pesticide runoff losses. A summary of the runoff losses of pesticides and the effectiveness of two SWCPs in reducing these losses is given in Table 9-11. The table indicates that the amount of pesticide carried in runoff usually consists of only a small fraction of the total amount of material applied. The small fraction which is carried in runoff can be effectively reduced by terraces with contours because terracing reduces the amount of runoff water as well as the amount of sediment transport. Less expensive, non-structural SWCPs appear to be less effective in reducing the transport of moderately adsorbed pesticides but are quite effective in reducing the transport of the few strongly adsorbed pesticides currently in use. It is possible that in some circumstances SWCPs might increase the amounts of pesticide leaving target areas through leaching; however, in most cases these losses would be relatively minor in comparison with reductions in runoff losses.

Pesticides differ from sediment in that their primary transport route away from target areas is often by drift or volatilization. In cases where pesticides are applied by aerial equipment, the amount of pesticide transported to non-target areas by air may be as much as several hundred times as large as the amount transported by runoff. Unfortunately, an evaluation of the relative impact of air-borne and runoff losses on water quality is not possible because studies have not been conducted to measure actual inputs of air-borne and water-borne pollutants to surface waters. However, in those cases where air-borne losses are almost 100 times larger than runoff losses, the impact of alternative pollution control methods on air-borne losses as well as on water borne losses should be considered.

In the cases of methyl parathion and toxaphene, which are applied primarily by aerial equipment, the amounts of toxic material lost in drift can be as much as two orders of magnitude greater than the amounts lost in runoff. Because of these large air-borne losses to non-target areas, changing application procedures by replacing aerial applications with ground applications would be more effective at reducing methyl parathion and toxaphene transport than would SWCPs. However, improvements in pesticide efficiency through the implementation of IPM programs seem to be the most cost-effective means of preventing the contamination of non-target areas by toxaphene and methyl parathion since both air-borne and water-borne losses are reduced with improved pest control methods that usually have no net cost to the farmer.

The most effective method for stopping paraquat transport is to eliminate its use. In many cases, this can be done by replacing no-till corn with an alternative tillage practice. SWCPs are also quite effective at controlling the amount of paraquat lost with eroded sediment; however, because this material is so tightly bound under most soil conditions, paraquat leaving agricultural fields in runoff does not appear to pose any dangers to aquatic systems and thus does not need to be controlled. On the other hand, the air-borne losses of paraquat are very hazardous and should be controlled

TABLE 9-11. REDUCTIONS IN RUNOFF LOSSES WITH VARIOUS POLLUTION CONTROL STRATEGIES

Pesticide	Type of Study	Application Rate	Runoff Losses	Runoff Losses With SWCP		Runoff Losses With IPM Programs [1]
				Contours	Terraces With Contours	
Methyl Parathion	Field	13400 g/ha	0.008 - 0.25% (1.1 - 33.5 g/ha)	0.007 - 0.21% [2] (0.91 - 27.8 g/ha)	0.004 - 0.125% [2] (0.55 - 16.75 g/ha)	0.0043 - 0.135% (0.58 - 18.1 g/ha)
	Modelling	6720 g/ha	4.6% (310.7 g/ha)	3.8% (256.5 g/ha)	2.3% 154.7 g/ha	
Toxaphene	Field	26900 g/ha	0.078 - 0.72% (21 - 193.7 g/ha)			0.048 -0.44% (12.91 - 118.4 g/ha)
	Modelling			~0.052 - 0.48% [3] (~14 - 127.9 g/ha)	~0.037 - 0.35% [3] (10.1 - 93 g/ha)	
Carbofuran	Field	~1120 g/ha	0.05 - 2.0% (0.56 - 22 g/ha)			0.025 - 1.0% (0.28 - 11 g/ha)
	Modelling			~0.04 - 1.6% [4] (0.45 - 17.6 g/ha)	~0.02 - 0.8% [4] (~0.22 - 8.8 g/ha)	
Atrazine	Field	~2910 g/ha	1.0 - 16% (29.1 - 465.6 g/ha)	0.76 - 12.16% [2] (23.12 - 353.9 g/ha)	0.36 - 5.8 [2] (10.5 - 167.6 g/ha)	
	Modelling	2910 g/ha	1.8% (53.5 g/ha)	1.4% (40.9 g/ha)	0.65% (19.0 g/ha)	
Paraquat	Field	1120 g/ha	1.28% (14.3 g/ha)	0.85% [2] (9.4 g/ha)	0.62% [2] (6.9 g/ha)	
	Modelling	2520 g/ha	13.5% (340.6 g/ha)	8.9% (225 g/ha)	6.4% (162.5 g/ha)	

[1] Assumes that reductions in runoff losses are equal to reductions in rate of application.

[2] Estimates of SWCP reductions in pesticide runoff losses made by using modelling results for SWCPs and runoff losses of pesticides in empirical studies.

[3] Estimates based on reductions in paraquat's runoff losses shown in modelling study of Beyerlein and Donigian (Appendix F) and runoff losses of toxaphene seen in empirical study.

[4] Estimates based on runoff losses of carbofuran seen in empirical study and modelling results of Beyerlein and Donigian (Appendix F for atrazine and water losses in runoff.

Beyerlein and Donigian study found:
·reductions of atrazine in runoff with contours = -24%
·reductions in water runoff losses with contours = -11%
·reductions of atrazine in runoff with terraces with contours - -64%
·reductions in water runoff losses with terraces with contours = -36%

Thus, we estimated that carbofuran would have 20% reductions in runoff losses with contours, and 60% reductions when terraces with contours are used.

especially in those areas where paraquat is applied by aerial equipment.

Atrazine differs from paraquat in that its losses in runoff are available to biological organisms, thus the control of these losses can be beneficial to the environment. Programs to increase the efficiency of atrazine use have not been developed at this point in time; hence, SWCPs like stripcropping and terraces with contours may prove to be the best pollution control methods available. Substitution with less persistent herbicidal materials will also reduce the runoff losses of toxic materials from agricultural fields (see Section 8). As mentioned previously, the losses of atrazine are quite small except when the material is applied by ground equipment in windy conditions and when the material is applied by aerial equipment. Volatilization losses can be larger than either the drift or runoff losses of atrazine; however, little is known about the relative hazards that drift, volatilization, and runoff losses of atrazine pose for water quality.

Because carbofuran is usually applied in a granular formulation with ground equipment and because its volatilization losses are insignificant, carbofuran has the lowest air-borne losses of all the pesticides examined. As a result, the major transport mode of carbofuran is runoff; thus, SWCPs may have a significant impact on the total amount of carbofuran leaving crop fields. The only field data on SWCPs and their impact on carbofuran's runoff losses showed that the runoff losses of carbofuran from no-till and conservation tillage corn grown on claypan soils were actually slightly higher than the runoff losses observed with conventional tillage. However, the Hydrocomp model (Appendix F) predicts that the use of terraces and contours would result in a substantial reduction in the movement of atrazine, a pesticide which would be expected to behave in a manner similar to carbofuran. Although terraces with contours are probably effective at preventing runoff of carbofuran, they are also usually quite expensive in comparison with other methods for reducing carbofuran's losses in runoff. The most effective of these alternative methods for reducing transport eliminates the need for carbofuran applications by rotating corn with an appropriate crop such as soybeans. In many areas such a rotation is quite profitable. In those areas where it is not, the use of insect population monitoring techniques to determine when applications are needed for corn rootworm shows considerable promise for substantially reducing the amount of carbofuran applied. Crop rotation and scouting appear to be as effective as terraces and contours in reducing carbofuran transport and are usually much less expensive.

The results of the analysis of alternatives for each of the representative pesticides are summarized in Table 9-12. The qualitative ratings are based upon the impact of each method on total transport of the material. In order to make these calculations it was necessary to assume that a reduction in application rate would be accompanied by a proportional reduction in runoff and air-borne losses from the target area. From Table 9-12 and the preceding discussion it appears that atrazine is the only one of the representative pesticides for which SWCPs might possibly be the most cost-effective means of preventing environmental contamination.

Our analysis of the advantages of various strategies for reducing environmental contamination from pesticides has been based on an analysis of

TABLE 9-12. SUMMARY EVALUATION

	Overall Environmental Rating[1]	Reduction of Toxic Material Entering the Environment[2]		
		With SWCP	With IPM	With Changes in Application Procedures
Atrazine	6	poor	---	very poor to fair
Carbofuran	12.0	very poor to poor	fair to excellent	very poor
Methyl Parathion	9.7	very poor to poor	excellent	fair to good
Paraquat	8.0	very poor[3]	---	very poor to fair
Toxaphene	10.0	very poor	excellent	fair to good

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[1] The numbers represent the value of an index of environmental hazard developed by Metcalf (1975). The index is based upon both toxicity and persistence data. The values for the insecticides are directly from Metcalf (1975). The values for the herbicides were computed by the same method. The values for insecticides in Metcalf's study ranged from a low of 4.3 for Trichlorfon to the most hazardous rating of 14.0 for Endrin.

[2] The evaluation is based on the average reduction as a percentage of the material originally applied. Very poor corresponds to a reduction of < 2%, poor to a reduction of < 5%, fair to a reduction of < 15%, good to a reduction of between 15% and 25% and excellent to a reduction of over 25%.

[3] Low rating is because the paraquat in sediment appears to lose its toxicity.

the amounts and toxicities of the materials transported by air and water. However, there is considerable uncertainty about both the amounts of material transported and their toxic impact on the environment. The uncertainty about the amounts transported is due not only to the lack of extensive monitoring but also to the limitations of the monitoring techniques themselves. For example, using ^{14} carbon labelled insecticide - Lichtenstein et al. (1977) discovered bound residues of methyl parathion that currently available monitoring techniques cannot detect. Similarly, the impact of pesticides on the aquatic and terrestrial environments is very difficult to assess: adsorption and desorption phenomena constantly change concentrations of toxic materials in water and soil, and sublethal, secondary and synergistic effects of pesticides on public health and aquatic and terrestrial ecosystems are not well understood. An example of these sublethal effects are the psychiatric disorders associated with exposure to organophosphorus insecticides. Studies establishing the mutagenicity of certain pesticides are equally disturbing.

Thus, given the limitations of pollution monitoring techniques and a multitude of unanswered questions about the sublethal, secondary and synergistic toxicities of the pesticides presently used in agriculture, it appears that pollution control methods which reduce the use of pesticides have a considerable advantage over methods which reduce the movement of pesticides that have already been released into the environment. Methods which decrease the usage of pesticides greatly reduce the potential for pollution events: less pesticide is available for the occurrence of runoff losses, air-borne transport, human and wildlife poisonings, accidental spills, and improper disposal. In certain cases, integrated pest management programs which reduce pesticide use may also reduce the energy needs and production costs of crop production. For those situations where such a reduction in pesticide use is not possible, prophylactic methods like SWCPs may prove to be the best way of containing toxic pesticide materials within agricultural areas. However, there is a great potential for improving the efficiency of pesticide use in U.S. agriculture thereby reducing the amounts of pesticide required for pest control; thus, emphasis should be placed on the development and implementation of programs which utilize biological and cultural pest control methods and improve the placement, timing and equipment used in pesticide application.

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APPENDIX A

CORNELL NUTRIENT SIMULATION (CNS) MODEL

This appendix summarizes the mathematical details of the simulation model used to estimate the effects of SWCPs on water, soil and nutrient losses at locations in New York, Iowa and Georgia. Parameters used in model validations for New York and Georgia are also presented. The simulation model consists of daily water balance and soil loss components and monthly nutrient inventories.

WATER BALANCE

The hydrologic component of the simulation model is based on a daily moisture balance for the top 30 cm of soil:

$$SW_{t+1} = SW_t + M_t + P_t - ET_t - E_t - R_t - L_t \qquad (A.1)$$

where SW_t = soil water content of the top 30 cm of soil at the beginning of day t(cm)

M_t = snowmelt on day t(cm)

P_t = precipitation as rain on day t(cm)

ET_t = evapotranspiration from the soil on day t(cm)

E_t = direct evaporation from the soil surface on day t(cm)

R_t = direct runoff, including interflow, on day t(cm)

L_t = leaching or deep percolation from the 30 cm surface soil layer on day t(cm)

The soil water content is constrained to be above the wilting point WP, (cm), and is assumed to drain to field capacity FC, (cm), unless the soil is frozen, in which case moisture content may reach saturation SAT, (cm).

Evapotranspiration during the growing season is assumed equal to potential evapotranspiration as estimated by the formula developed by Hamon (1961) provided soil water content exceeds the wilting point. Evaporation from bare soil or snowpack is assumed to be temperature-dependent and is determined from the equation developed by Eggleston _et al._ (1971). Snowmelt is determined using melt equations for clear or rainy days developed by the U.S. Army Corps of Engineers (1960).

Runoff

During periods of the year when the soil is neither frozen nor covered with snow, direct runoff is predicted using the U.S. Soil Conservation Service's curve number equation (Mockus, 1972; Ogrosky and Mockus, 1964):

$$R_t = \frac{(P_t - 0.2S_t)^2}{P_t + 0.8S_t} \qquad \text{for } P_t > 0.2S_t \qquad (A.2)$$

$$\text{and } R_t = 0 \qquad \text{for } P_t \leq 0.2S_t$$

where S_t is a runoff detention parameter for day t(cm). Values of S_t are determined from curve numbers which are defined as functions of soil hydrologic group, crop, supporting practice, hydrologic condition and antecedent moisture condition (AMC). The latter is usually determined using the five-day antecedent rainfall. Since this simulation model computes soil water content SW_t, antecedent moisture conditions can be determined directly from SW_t, provided soil moisture limits can be associated with the appropriate AMC categories: I(driest), II(average), and III(wettest). These three levels correspond to decreasing infiltration rates and increasing runoff. The soil moisture limits used in the simulation model are given in Table A-1. These limits were developed in such fashion that they are analogous to AMC categories based on 5-day antecedent rainfall. Since the difference in antecedent rainfall for AMC I and AMC III is in the order of 2 cm, it is assumed that the soil moisture limits should reflect a comparable difference. The determination of soil moisture for the driest category is based on the knowledge that when soil moisture falls below field capacity (FC), the soil macropores are open, and infiltration rates should be high. The AMC I limit is thus arbitrarily set to one centimeter below [illegible]

TABLE A-1. RELATIONSHIP BETWEEN SOIL WATER CONTENT AND ANTECEDENT MOISTURE CONDITION

Soil Water Content, SW_t(cm)	Antecedent Moisture Condition, AMC
$SW_t < FC-1.0$	I
$FC-1.0 \leq SW_t \leq FC+1.0$	II
$SW_t > FC+1.0$	III

The curve number equation was not developed to predict snowmelt runoff. Frozen soil will impede infiltration and percolation and hence increase runoff over summer conditions. Since the curve number equation is one of the few practical means for estimating runoff, it was adapted to snowmelt in the current study by modification of the detention parameter S_t. At least in humid areas, soil water content is at or exceeds field capacity during most snowmelt events, and the modification assumes that at $SW_t = FC$ when the soil is frozen, a modified detention parameter S'_t is equal to $0.5S(III)_t$, where $S(III)_t$ is the unmodified detention parameter for the wettest AMC. Frozen soil can have a moisture content exceeding field capacity, and when saturation is reached, any further precipitation or snowmelt will all become runoff. Assuming a linear relationship between field capacity and saturation (SAT, in cm), the following equation estimates the modified detention parameter.

$$S'_t = 0.5(1 - \frac{SW_t}{SAT})(\frac{SAT}{SAT - FC})\ S(III)_t \qquad (A.3)$$

where S'_t is a detention parameter for frozen soil(cm). The runoff equation for frozen soil becomes

$$R_t = \frac{(P_t + M_t - 0.2S'_t)^2}{P_t + M_t + 0.8S'_t} \qquad \text{for } P_t + M_t > 0.2S'_t \qquad (A.4)$$

$$R_t = 0 \qquad \text{for } P_t + M_t \leq 0.2S'_t$$

Whenever the soil is thawed, runoff is predicted using Equation (A.2) substituting $P_t + M_t$ for P_t.

Percolation

Percolation, or leaching occurs whenever soil moisture level exceeds field capacity. Drawing the analogy to five-day antecedent rainfall, the maximum daily percolation rate is assumed to be equal to (SAT - FC)/5.

SOIL EROSION

Soil loss is estimated from the modification of the universal soil loss equation proposed by Onstad and Foster (1975).

$$SL_t = 2.24W_tK(LS)C_tP \qquad (A.5)$$

where SL_t = soil loss on day t(MT/ha)

W_t = rainstorm kinetic energy

C_t = cover factor, day t

LS = topographic factor

P = conservation practice factor

K = soil erodibility factor

The parameters C_t, LS, P and K are the standard tabulated factors used in the universal soil loss equation. However, the energy term, W_t is computed for the rain falling on day t as follows:

$$W_t = 0.5E_t(\frac{I_t^{30}}{2.54}) + 15(\frac{R_t}{2.54})(\frac{q_t}{2.54})^{1/3} \qquad (A.6)$$

and

$$E_t = \left[916 + 331 \log(\frac{\overline{I}_t}{2.54})\right] \frac{P_t}{2.54} \qquad (A.7)$$

where I_t^{30} = the maximum 30-min rainfall intensity(cm/hr)

q_t = peak runoff(cm/hr)

$\overline{I}_t$ = average storm rainfall intensity(cm/hr)

To use the soil loss equations for long simulation periods, it is essential to estimate I_t^{30}, q_t and $\overline{I}_t$ from standard meteorological data (hourly rainfall). Hence, these values are determined from IC_t^{60}, the maximum 60-minute clock rainfall intensity (cm/hr) and storm duration D_t(hr). These three required factors are computed as follows

Average storm rainfall intensity is given by

$$\overline{I}_t = P_t/D_t \qquad (A.8)$$

Maximum 60-min intensity I_t^{60}, is approximately 1.13 IC_t^{60} (Hershfield, 1961). As indicated in Schwab et al. (1966, p. 624),

$$I_t^{30} = 1.58\ I_t^{60} \qquad (A.9)$$

Assuming that storm duration is greater than the field's time of concentration, T_c(hr), peak runoff is given by

$$q_t = (I_t^c - f_t) \qquad (A.10)$$

where I_t^c is the average rainfall intensity during the time in which T_c is reached (cm/hr) and f_t is the infiltration rate.

To compute I_t^c, and estimate must be made based on I_t^{60}. The graphical relationship given in Schwab et al. (1966, p. 624) can be approximated as

$$I_t^c = \begin{cases} \dfrac{1.34T_c^{0.62}I_t^{60}}{T_c} & T_c \leq 1/3 \text{ hr} \\ \dfrac{T_c^{0.34}I_t^{60}}{T_c} & T_c > 1/3 \text{ hr} \end{cases} \tag{A.11}$$

where T_c is given by Schwab _et al._ (1966, p. 629) as

$$T_c = \frac{1.67(3.28\ell)^{0.8}(S_t/2.54 + 1)^{1.67}}{9000s^{0.5}} \tag{A.12}$$

In equation 12 ℓ and s are slope length and gradient, respectively, in meters and percent. As a final step, the infiltration rate f_t(cm/hr) is estimated as the minimum rates for the relevant soil category (Musgrave and Holton, 1964): A: 0.75-1.15cm/hr, B: 0.40-0.75cm/hr, C: 0.15-0.40cm/hr, D: 0-0.15cm/hr.

The soil loss equation A.5 is applicable to rainfall erosion only. In northern climates, erosion associated with snowmelt is neglected.

NUTRIENT BALANCE

While it is apparent that daily soil water levels vary sufficiently that reasonable runoff predictions can be obtained only from a daily model, soil nutrient levels can be adequately modelled on a monthly basis. Moreover, since the dates of fertilizer and manure applications are often uncertain, there is little point in the 30-fold increase in computations required by a daily over a monthly nutrient model. The nutrient balance model estimates monthly losses of dissolved and solid-phase nitrogen (N) in runoff, dissolved N in percolation, dissolved available (resin - extractable) phosphorus (P) in runoff and solid-phase P in runoff. It is assumed that all dissolved nutrients are in the inorganic form and that solid-phase N is organic. Losses of solid-phase nutrients are assumed to be fixed or adsorbed to soil particles. The monthly runoff, percolation and soil loss values required for the nutrient balances are obtained by summing the daily values predicted by the water balance and erosion components of the simulation model.

Nitrogen

The soil nitrogen budget is described by inventory equations for soil inorganic and organic nitrogen.

$$NI_{n+1} = NI_n + m_n NO_n + MN_n + PI_n - RI_n - LN_n - CN_n + FN_n \tag{A.13}$$

$$NO_{n+1} = NO_n(1-m_n) - RO_n + MNO_n \tag{A.14}$$

where NI_n = soil inorganic nitrogen content at the beginning of month in (kg/ha)

NO_n = soil organic nitrogen content at the beginning of month n (kg/ha)

m_n = fraction of soil organic nitrogen mineralized to inorganic nitrogen during month n

MN_n = inorganic nitrogen added to the soil from manure during month n(kg/ha)

PI_n = precipitation additions of inorganic nitrogen in month n(kg/ha)

RI_n = inorganic nitrogen lost in runoff during month n(kg/ha)

LN_n = inorganic nitrogen lost in percolation (leaching) during month n(kg/ha)

CN_n = crop uptake of inorganic nitrogen during month n(kg/ha)

FN_n = fertilizer applications of inorganic nitrogen during month n(kg/ha)

RO_n = organic nitrogen lost in eroded soil during month n(kg/ha)

MNO_n = addition to soil organic nitrogen content from manure and/or fresh plant residues during month n(kg/ha)

The inorganic nitrogen budget is applied to the top 30 cm of soil as with the hydrologic model. The organic nitrogen budget is limited to the top 10 cm, since the microbial populations responsible for mineralization are most prevalent near the soil surface. When applied to legumes, the fertilizer nitrogen input, FN_n, must include nitrogen fixation inputs. The various chemical forms of inorganic nitrogen are lumped together in equation A.13, implying that oxidation to nitrate is rapid; i.e., most available soil inorganic nitrogen exists as nitrate.

Equation A.13 does not account for gaseous losses of soil inorganic nitrogen due to denitrification and ammonia volatilization. Such losses can be substantial, but they are very difficult to model. Although kinetic relationships can be developed for denitrification and ammonia volatilization, the appropriate rate constants for these processes cannot be directly estimated from soil properties. To partially account for gaseous losses in applications of the model, 25% of the fertilizer N inputs are substracted.

The soil organic nitrogen, NO_n, refers to the relatively stable organic nitrogen tied up in humic and other resistant material. Such nitrogen mineralizes slowly (2-3% annually) and must be distinguished from the organic nitrogen in fresh manure or plant residues which can mineralize more rapidly. Nitrogen transformations in manure which has been applied to the soil are very difficult to describe quantitatively and in this simulation model they are described in a rather arbitrary fashion as follows:

1. Based on the studies of Chichester _et al._ (1975) it is assumed that manure organic nitrogen mineralizes at three times the rate of soil organic nitrogen (i.e., $3m_n$) during the first year following application. In the second year this rate is reduced to $1.5m_n$ and thereafter it is assumed that manure organic nitrogen joins the soil organic nitrogen pool (MNO_n in equation A.14).

2. The addition of inorganic nitrogen from manure (MN_n in equation A.13 is the sum of the manure organic nitrogen mineralized during month n and the inorganic nitrogen in the manure applied during the month. The latter is adjusted for ammonia losses in surface spreading. Based on the work of Lauer _et al._ (1976), it is assumed that such losses are 85% of the ammonia content of the manure when spread.

Fresh plant residues are assumed to have mineralization properties similar to manure.

The mineralization rate, m_n is assumed temperature-limited, and can be modelled by the Van't Hoff-Arrhenius relationship (Haith, 1973). Within the typical range of soil temperatures (0-20°C), a linear approximation is possible (Haith _et al._ 1977):

$$m_n = m_o \frac{T_n}{\Sigma T_n} \qquad \text{for } T_n \geq 0 \qquad \text{(A.15)}$$

where m_o = yearly mineralization rate as a fraction of the average soil organic nitrogen content(dmless)

T_n = average air temperature in month n(°C)

Equation A.15 apportions the yearly mineralization to the various months on the basis of degree-months. Mineralization is assumed zero during any month in which the average air temperature is less than or equal to zero.

Precipitation inputs of inorganic nitrogen, PI_n, are relatively small and may be neglected or modelled as:

$$PI_n = 0.1 np_n P_n \qquad \text{(A.16)}$$

where np_n = concentration of inorganic nitrogen in precipitation in month n(mg/ℓ)

P_n = total precipitation in month n(cm)

Crop uptake of nitrogen is assumed to follow a sigmoid curve of seasonal crop growth. The curve is scaled by the growing season and total nitrogen uptake over the growing season.

The most important part of the nutrient model is the prediction of losses in runoff and percolation. These losses are modelled as the product of concentration and water volume, where concentration is determined from average nitrogen level in the soil during the month. The average concentration of inorganic nitrogen in percolation or leaching water L_n during month n is given by $\overline{NI}_n/(FC + R_n + L_n)$ where $\overline{NI}_n$ is the average soil inorganic nitrogen content in month n(kg/ha) and FC is field capacity. Average soil inorganic nitrogen is:

$$\overline{NI}_n = \frac{NI_n + NI_{n+1}}{2} \tag{A.17}$$

The inorganic nitrogen is mixed in water volume $FC + R_n + L_n$, since percolation occurs only after the soil reaches field capacity. Percolation losses of inorganic nitrogen are thus given by:

$$LN_n = L_n \frac{\overline{NI}_n}{FC + R_n + L_n} \tag{A.18}$$

A similar computation is made for losses of inorganic nitrogen in runoff.

$$RI_n = R_n \frac{\overline{NI}_n}{SAT + R_n + L_n} \tag{A.19}$$

The soil's saturation water content (SAT) is arbitrarily substituted for field capacity (FC), in determining nitrogen concentration. This substitution is made to produce the lower nitrogen concentration in runoff than in percolation that has been reported elsewhere (Biggar and Corey, 1970). It is rationalized by the observation that, when runoff occurs, the near-surface soil is saturated. This is not, however, the most satisfactory way to make the concentration adjustment. Lower nitrogen concentrations in runoff than in percolation are due to the fact that more soil nitrogen is exposed to percolation water than runoff. Thus, in equation A.19, $\overline{NI}_n$ should be replaced by some fraction of $\overline{NI}_n$. The fraction will depend on the distribution of runoff and interflow, and there appears to be no practical way to quantify this distribution. The curve number runoff equation predicts the sum of interflow and surface runoff and there have been very few field studies in which the two runoff components were measured separately.

The organic nitrogen lost with eroded soil RO_n as defined for equation A.14, is given by:

$$RO_n = \frac{SL_n \overline{NO}_n ERN}{WT_{10}} \tag{A.20}$$

where SL_n = average soil loss for month n(kg/ha)

$\overline{NO}_n$ = average organic nitrogen in the top 10 cm of soil (kg/ha)

ERN = enrichment ratio for organic nitrogen in eroded soil

WT_{10} = weight of the top 10 cm of soil

The organic nitrogen and soil weight terms in equation 20 are limited to the top 10 cm since this upper layer is the basis for the organic nitrogen mass balance (equation A.21). The average organic nitrogen content is given by:

$$\overline{NO}_n = \frac{NO_n + NO_{n+1}}{2} \qquad (A.21)$$

The enrichment ratio is typically between 2.0 and 4.0 (McElroy _et al_ 1976). A value of 2.5 is used in the simulation model.

Phosphorus

The reactions of phosphorus (P) in the soil-water system are substantially different from nitrogen reactions, and must be modelled according to a different set of assumptions. Unfortunately, organic P transformations in the soil are not well defined. The simulation model considers only inorganic P, and it is assumed that the breakdown of complex organic P compounds into inorganic P will not greatly affect the availability of P to crops or increase runoff losses of P. Furthermore, the phosphorus inventory equations consider only _available_ inorganic P (orthophosphate-P). Disregarding the transformations between organic and inorganic P and between fixed and available P essentially limits the simulation model's ability to predict P losses to the short term (2-3 years). The two basic inventory equations for available P are:

$$TP_n = PA_n + PD_n \qquad (A.22)$$

and

$$TP_{n+1} = TP_n + FP_n + MP_n - CP_n - RPD_n - LPD_n - RPA_n \qquad (A.23)$$

where TP_n = total available P in the surface soil layer at the beginning of month n(kg/ha)

PA_n = adsorbed available P in the surface soil layer at the beginning of month n(kg/ha)

PD_n = dissolved available P in the soil water at the beginning of month n(kg/ha)

FP_n = fertilizer inorganic phosphorus added to the soil during month n(kg/ha)

MP_n = manure inorganic phosphorus added to the soil during month n(kg/ha)

CP_n = phosphorus removed from the soil by plant uptake (kg/ha) during month n(kg/ha)

RPD_n = loss of dissolved available P in runoff during month n(kg/ha)

LPD_n = loss of dissolved available P in percolation during month n(kg/ha)

RPA_n = loss of adsorbed available P in eroded soil during month n(kg/ha)

Crop removal of soil phosphorus, CP_n, is determined in a fashion similar to crop nitrogen removals.

The equilibrium between adsorbed and soluble phosphorus is approximated by a linear isotherm:

$$pa_n = K_p(pd_n) \qquad (A.24)$$

where pa_n = adsorbed available P concentration in the surface soil layer at the beginning of month n(ppm; mg/kg)

pd_n = dissolved available P concentration in the surface soil water at the beginning of month n(ppm; mg/ℓ)

K_p = P adsorption coefficient

In terms of phosphorus contents in kg/ha, equation A.24 becomes:

$$PA_n = \frac{K_p(WT_{30})}{10^5 SW_n} PD_n \qquad (A.25)$$

where WT_{30} = weight of the top 30 cm of soil (kg/ha)

The equilibrium constant K_p is a function of soil properties. Data from Enfield and Bledsoe (1975) were used to develop the regression equations given in Table A-2. The independent variables used in the equations are pH and

TABLE A-2. PREDICTIVE EQUATIONS FOR PHOSPHORUS ADSORPTION COEFFICIENT

Number of Soils	Number of Soils Omitted	Regression	R^2
19	0	$K_p = 28.4 + 2.5(\%C) + 21.6(pH-6)^2$	0.42
18	1	$K_p = 1.5 + 3.0(\%C) + 24.4(pH-6)^2$	0.53
17	2	$K_p = 5.1 + 2.2(\%C) + 26.4(pH-6)^2$	0.62

percent clay (%C), defined as the soil particle percentage with diameters less than 0.002 mm. Enfield and Bledsoe equilibrated 25 soils at a variety of initial dissolved P concentrations. Equilibrium times ranged from one to 3000 hours. The regression equations given in Table A-2 were determined from Enfield and Bledsoe's experimental data for low (0.5-1.0 mg/ℓ) initial P concentrations and 300-hr equilibrium times. Six soils were analyzed destructively and 19 were analyzed nondestructively. For consistency, only results from the latter were used in the regressions. The three equations given in Table A-2 are based on elimination of zero, one or two outliers. Although the equations give similar results, elimination of the two most extreme soils significantly improves the fraction of variation explained (from 0.42 to 0.62), and thus the final equation,

$$K_p = 5.1 + 2.2(\%C) + 26.4(pH-6)^2 \qquad (A.26)$$

is used in the simulation model.

Runoff and percolation losses of available P are computed similarly to nitrogen losses:

$$RPD_n = R_n \frac{\overline{PD}_n}{SAT + R_n + L_n} \qquad (A.27)$$

$$LPD_n = L_n \frac{\overline{PD}_n}{FC + R_n + L_n} \qquad (A.28)$$

where $\overline{PD}_n$ = average dissolved available P in the soil during month n(kg/ha)

$$\overline{PD}_n = \frac{PD_n + PD_{n+1}}{2} \qquad (A.29)$$

The estimate of P percolation losses (LPD_n) is somewhat misleading, since the leached P is likely to be removed from percolation waters by adsorption in the subsoil well before reaching groundwater aquifers.

Losses of adsorbed available phosphorus RPA_n are given by:

$$RPA_n = \frac{SL_n \overline{PA}_n ERP}{WT_{30}} \qquad (A.30)$$

where WT_{30} = weight of the top 30 cm of soil (kg/ha)

ERP = enrichment ratio for adsorbed P in eroded soil

PA_n = average adsorbed available P in the surface soil during month n(kg/ha) and

$$\overline{PA}_n = \frac{PA_n + PA_{n+1}}{2} \qquad (A.31)$$

Total loss of solid phase phosphorus is the sum of both available and fixed P.

$$PP_n = \frac{SL_n Erp(\overline{PA}_n + PF)}{WT_{30}} \qquad (A.32)$$

where PF is the fixed (unavailable) phosphorus in the surface soil layer (kg/ha). As reported in McElroy et al. (1976), the phosphorus enrichment ratio is similar in magnitude to the ratio for organic nitrogen. A value of ERP = 2.0 is used in the simulation.

VALIDATION DATA

Georgia Validations

Principal data for the Watkinsville validations are given in Table A-3. Validation results are presented in Table 6-4. Crop curve numbers and cover factors (C_t) were changed according to the crop management changes summarized by Smith et al. (1978). Nutrient inputs and crop yields were obtained from this same source. Yields were subsequently used to estimate crop nutrient uptake.

New York Validations

Analogous data for the Aurora validations are given in Table A-4. Validation results are presented in Table 6-6. Crop curve numbers and cover factors were based on continuous corn, with or without residues left during winter months (corresponding to well and poorly managed plots). Nutrient inputs and crop yields were obtained from Klausner et al. (1976) and from unpublished data provided by Stuart D. Klausner, Department of Agronomy, Cornell University.

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TABLE A-3. PARAMETER VALUES USED FOR WATKINSVILLE, GA. MODEL VALIDATION

Parameter	Value			
	P1	P2	P3	P4
SAT	11.7 cm			
FC	8.4 cm			
WP	4.2 cm			
f_t	0.6 cm/hr			
K	0.28			
m_o	0.03			
WT_{10}	1.65 10^6kg/ha			
WT_{30}	3 WT_{10}			
P	0.5	0.6	0.5	0.5
ℓ	100m	61m	26m	26m
s	4%	2%	3%	3%
LS	0.64	0.25	0.27	0.27
NI_1	---	5 kg/ha	---	5 kg/ha
NO_1	---	580 "	---	790 "
TP_1	---	150 "	---	115 "
PF	---	750 "	---	690 "
pH	---	5.8	---	6.1
%C	---	17.0%	---	22.5%

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TABLE A-4. PARAMETER VALUES USED FOR AURORA, N.Y. MODEL VALIDATIONS

Parameter	Value
SAT	15.0 cm
FC	9.8 cm
WP	4.9 cm
NI_1	15 kg/ha
NO_1	1610-3770 kg/ha
m_o	0.02
WT_{10}	1.31 10^6 kg/ha
TP_1	16-90 kg/ha
PF	3100 kg/ha
%C	11.1
pH	6.7
WT_{30}	$3WT_{10}$

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APPENDIX B

THE CORNELL PESTICIDE MODEL (CPM)

This appendix summarizes the mathematical details of the simulation model which evaluates the interaction of water, soil and pesticide characteristics as they affect pesticide losses in surface and leaching water. Parameters used as input for model validation are also presented.

The simulation model is event based. Water and pesticide balances, and soil loss components are updated when a rain day or root zone depth change occurs. Downward movement of pesticides is calculated only after a significant quantity of water has infiltrated into the soil.

The soil profile is divided into four layers in this model (see Figure 7-2). Zone 1 is the first 5 cm of top soil. Zone 2 includes the mass of the roots and extends from 5 cm depth to the bottom of the active rooting zone. Zone 3 denotes the soil below the active rooting zone and above the maximum depth of rooting. Zone 4 is the remaining soil down to the impermeable layer.

Figure B-1 shows the development of the active root zone for corn at various locations in the United States. Step functions which simulate the change in the size of the active rooting zone through the year were based on the curves shown for corn. These functions are shown in Figure B-2. The variation between locations was due to climate and depth to impermeable layer.

THE SUB-MODELS

The simulation model consists of 5 sub-models, each of which reads in data from tape and records results on tape (see Figure 7-1). This structure was chosen so that model components could be changed easily, and so that experimental findings could be substituted for indirectly determined input data.

Sub-model TEMPMELT

Event Based Data Set--

The soil temperature and snowmelt sub-model first reduces the input daily weather data set to event based data set. Information is filed only for those days on which 0.01 cm or more of snowmelt or rainfall occurs. Snowmelt is determined by using melt equations developed by the U.S. Army Corps of Engineers (1960).

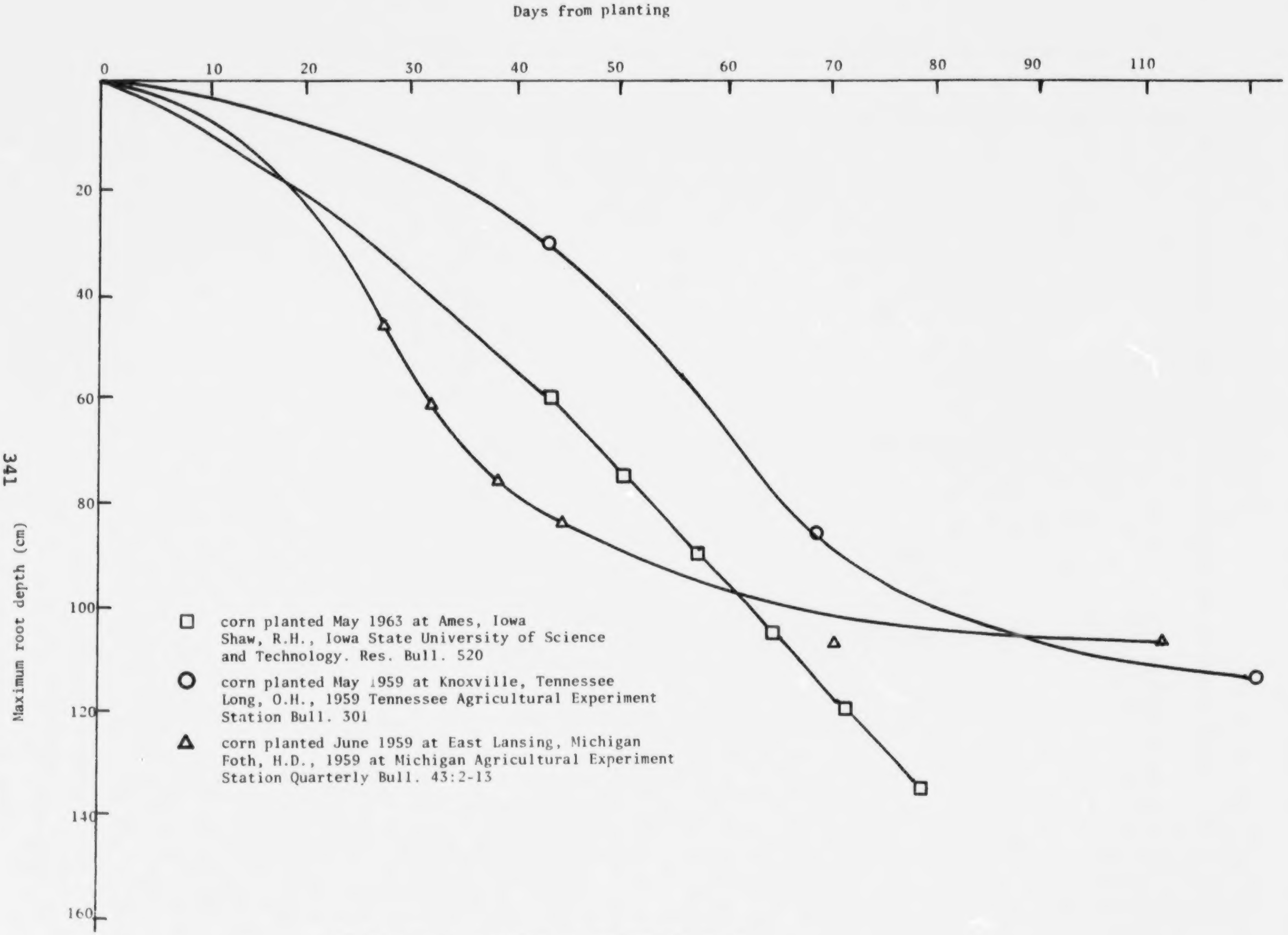

FIGURE B-1- LENGTH OF ROOTS OF CORN.

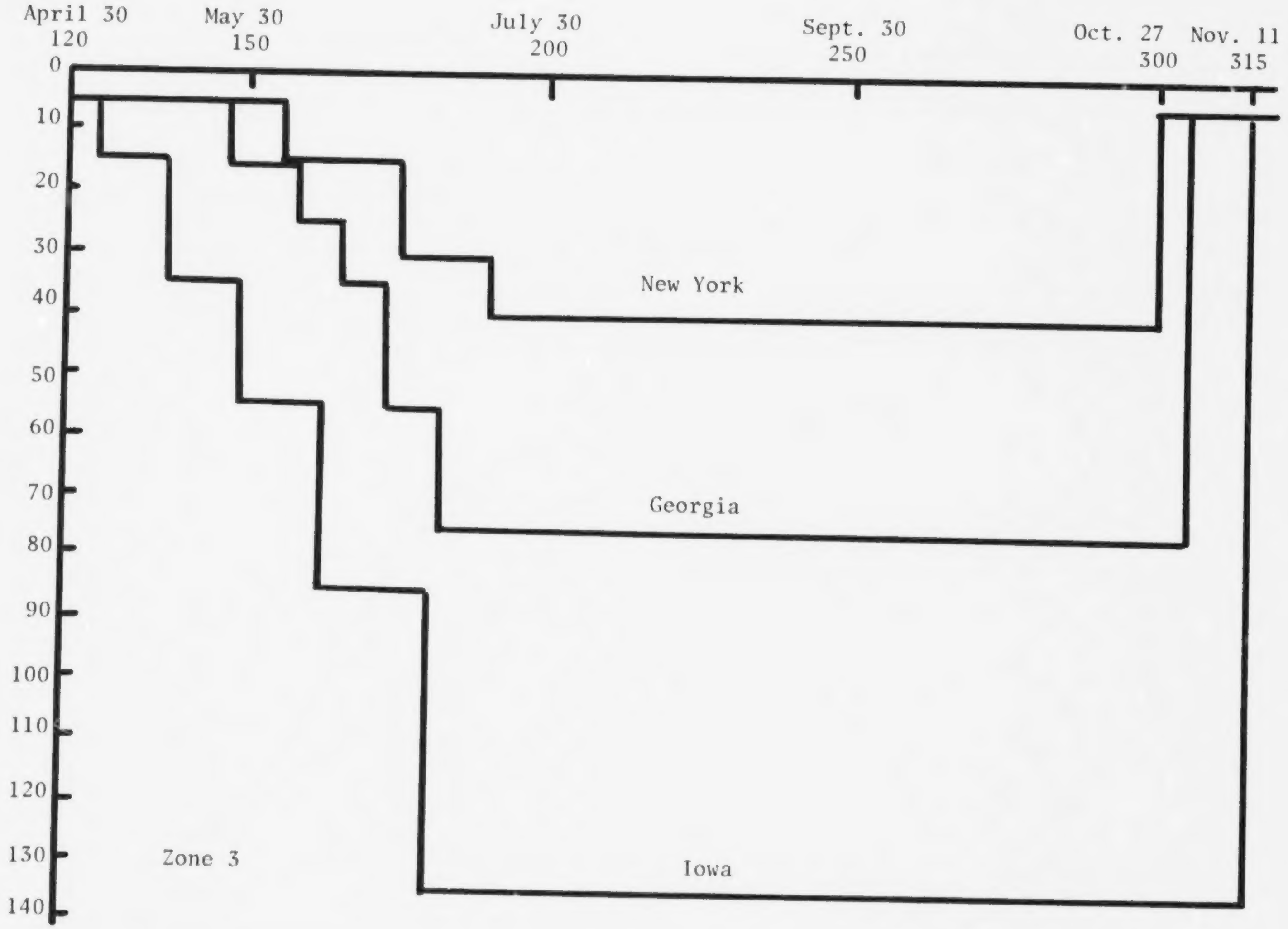

FIGURE B-2. LOWER BOUNDARY OF ROOT ZONES IN THE CORNELL PESTICIDE SIMULATIONS

Soil Temperature--

Next, a preliminary value for soil temperature is calculated as the running average of the previous 20 days' air temperature. Here, the observed value of air temperature is used except when there is snow accumulated on the ground; for this condition, the insulating property of snow is taken into account. On days when the water equivalent of snow exceeds 1.0 cm, air temperature is adjusted empirically as follows:

$$\text{ADJAIRTEMP} = \frac{\text{AIRTEMP}}{10 \cdot \text{WATEREQ}} \qquad \text{(B.1)}$$

where ADJAIRTEMP is the adjusted air temperature, °C

AIRTEMP is the actual air temperature, °C

WATEREQ is the water equivalent of snow, cm

To develop a predictive equation, preliminary soil temperatures were then regressed with soil temperatures observed at 10 cm depth. Observations are not available for all years of the simulation. Regression equations for New York and Iowa are:

New York: $\text{SOILTEMP} = 1.53 + 0.98 \cdot \text{ADJAIRTEMP} \quad (r^2 = 93\%)$ (B.2)

Iowa: $\text{SOILTEMP} = 0.52 + 0.84 \cdot \text{ADJAIRTEMP} \quad (r^2 = 90\%)$ (B.3)

Hydrological Sub-model

Moisture Budget--

The hydrological sub-model budgets soil moisture for each of the top three zones in the soil profile as follows:

$$SW(1)_t = SW(1)_{t-\Delta t} + M_t + P_t - ET(1)_{\Delta t} - R_t - \text{FLOW12} - \text{INTCEP} \qquad \text{(B.4)}$$

$$SW(2)_t = SW(2)_{t-\Delta t} + \text{FLOW12} - \text{FLOW23} - ET(2)_{\Delta t} \qquad \text{(B.5)}$$

$$SW(3)_t = SW(3)_{t-\Delta t} + \text{FLOW23} - \text{FLOW34} \qquad \text{(B.6)}$$

where $SW(J)_t$ = soil water content of Zone J at the end of day t, cm

Δt = the number of days elapsed since the last rain or snowmelt event

M_t = snowmelt on day t, cm

P_t = rainfall on day t, cm

$ET(J)_{\Delta t}$ = evapotranspiration from Zone J in the period day $(t-\Delta t)$ through day t, cm

R_t = runoff on day t, cm

FLOWIJ =water flow from Zone I to Zone J, cm

INTCEP =interception of rain water by plants, cm

Soil Water Movement--

In deep well-drained soils with moisture contents above field capacity, water moves readily under gravitational force from one zone to the next when the moisture content is above field capacity. For the purposes of simplicity, it is assumed that water does not move downward when the water content is below field capacity. Furthermore, when the water content reaches the wilting point, water is no longer available for plant uptake.

Runoff--

During periods of the year when the soil is neither frozen nor covered with snow, the partition of rainfall into an infiltration and a runoff component may be accomplished by one of three methods. The first method is based on the SCS Curve Number equation and is similar to the method used in the Cornell Nutrient Simulation Model (CNS). Appendix A gives details of these methods.

The second method is based on the Green and Ampt infiltration equation (Green and Ampt, 1911). This equation assumes a sharp wetting front, a constant hydraulic conductivity in the wetted zone, and a constant negative water pressure at the wetting front. The Green and Ampt equation can be expressed as:

$$i = k\,\frac{hw + z}{z} \tag{B.7}$$

where i = infiltration rate, cm/hour

k = hydraulic conductivity of the soil in wetted zone, cm/hour

hw = suction head at wetting front, cm

z = depth of wetting front, cm

Values for the saturated hydraulic conductivity and condition of wetting front for the soil groups are given in Table B-1.

Extensions of the Green and Ampt concept by Bouwer (1976) and Mein and Larson (1971) assume continuous ponding of water on the soil surface. They are, respecitively,

$$I = f \cdot z \tag{B.8}$$

and

$$t = \frac{f}{k} \cdot [z-[hw \cdot \ln (\frac{hw + z}{hw})]] \tag{B.9}$$

where I = the amount of water which has infiltrated into a uniform soil when the wetting front is at depth z, cm f = the difference between the volumetric water content above and below the wetting front; also called the fillable pore space, cm^3/cm^3

t = the time required for the wetting front to reach depth z under continuous ponding, hr

TABLE B-1. HYDRAULIC CONDUCTIVITY AND SUCTION AT WETTING POINT

Hydraulic soil group	Hydraulic conductivity* (cm/hr)	Suction at wetting front** (cm)
A	0.9	5
B	0.6	22.5
C	0.3	10
D	0.1	7

* England (1970)
**Compiled from data of Li _et_ _al_. (1976)

If rainfall intensity varies, ponding is not continuous. In this case, the Green and Ampt equation is modified as follows:

1) The rainfall event is divided into periods with constant intensities

2) For each such period, an effective rainfall intensity is computed to take into account the water stored in the surface depressions

Thus,

$$p_{eff} = \frac{p \cdot t_{per} + ST}{t_{per}} \qquad (B.10)$$

where p_{eff} = effective rainfall intensity, cm/hr

p = rainfall intensity, cm/hr

t_{per} = duration of sub-period, hr

ST = water stored in depressions, cm

3) The depth of the wetting front at the end of the period is found by assuming that all the rain and ponded water has infiltrated during this period. Thus,

$$z^* = z_b + p_{eff} \cdot (t_{per}/f) \qquad \text{(B.11)}$$

where z^* = hypothetical depth of wetting front if all water would have been infiltrated, cm

z_b = depth of wetting front at the start of the period, cm

As the wetting front crosses a soil zone boundary, a weighted average of the f's of all the two zones is substituted for f in Equation 11.

4) The infiltration rates at the beginning (i_b) and the end (i^*_{end}) of the period are found by using Equation 7:

$$i_b = \frac{hw + z_b}{z_b} \qquad \text{(B.12)}$$

$$i^*_{end} = k\,\frac{hw + z^*}{z^*} \qquad \text{(B.13)}$$

5) The computations of surface runoff, water stored in depressions, and depth of wetting front at the end of the period depend on the relative magnitudes of i_b, i^*_{end} and p_{eff}.

5a) When the effective rainfall intensity (p_{eff}) is smaller than the soil infiltration rate at the end of the period (i^*_{end}), there is no surface storage at the end of the period, no runoff generated during the period, and the wetting front will reach the maximum possible depth for that period, z^*. In symbol form,

$$z = z^* \qquad \text{(B.14)}$$

$$R = 0 \qquad \text{(B.15)}$$

$$ST = 0 \qquad \text{(B.16)}$$

where R is runoff (cm)

5b) If the infitration rate at the beginning of the period (t_b) is less than or equal to the rainfall intensity, then water will pond or run off. Then the depth of the wetting front at the end of the period, z_{end}, computed using a variation of Equation B.9 derived from Mein and Larson (1974):

$$t_{per} = \frac{f}{k}\left[(z_{end} - z_b) - \left[hw \cdot \ln\left(\frac{hw + z_{end}}{hw + z_b}\right)\right]\right] \qquad \text{(B.17)}$$

$$t_{per} = t_b - t_{end} \qquad \text{(B.18)}$$

This equation cannot be solved directly for z_{end}. Rather, the computer program uses an iterative process. As before, the f is adjusted when a soil zone boundary is passed.

The runoff during the period can be found as:

$$R = t_{per} \cdot p - STmax + ST_b \tag{B.19}$$

where STmax = the maximum quantity of water which can be stored in depressions, cm

ST_b = depression in storage at beginning of the period, cm

If the runoff computed by Equation B.19 is negative, the surface storage is

$$ST_{end} = ST_b = p \cdot t_{per} \tag{B.20}$$

and the runoff is set to 0. Otherwise,

$$ST_{end} = STmax \tag{B.21}$$

5c) Finally, if the effective rainfall intensity (p_{eff}) is smaller than the infiltration rate at the beginning of the period (i_b) but larger than the infiltration rate at the end of the period (i^*_{end}), then sometime during the period, the p_{eff} will become equal to the infiltration rate.

The wetting front depth when this occurs, z_{po} is:

$$z_{po} = \frac{hw \cdot k}{p_{eff} - k} \tag{B.22}$$

The time elapsed from the beginning of the period to reach z_{po} can be found as

$$t_{po} = f\left(\frac{z_b - z_{po}}{i_{eff}}\right) - t_b \tag{B.23}$$

and the surface storage at that time is

$$ST_{po} = ST_b - t_{po}\,(p_{eff} - p) \tag{B.24}$$

Hereafter, the surface runoff wetting front, and surface at end of period can be found with equations B.17 - B.21 with these substitutions:

$$t_b = t_b^{old} - t_{po} \tag{B.25}$$

$$ST_b = ST_{po} \tag{B.26}$$

$$z_b = z_{po} \tag{B.27}$$

where t_b = new value of t_b

t_b^{old} = previous value of t_b

The third method for finding runoff is used only for shallow soils with hardpans within 50 cm of the surface. This method assumes that runoff occurs only after the soil is completely saturated. Water flow in fragipan soils is mainly horizontal. Therefore, this flow is slower than the flow in deep drained soils. The maximum flow, "MASFLRATE", rate in fragipan soils is:

$$\text{MAXFLRATE} = \frac{(\text{SAT} - \text{FC}) \cdot \text{DEPIMP}}{20} \qquad (\text{B.28})$$

Water does not flow when soil moisture content is below field capacity.

Infiltration in Frozen Soils--

Water infiltration decreases as soil temperature decreases, when the water in the soil is frozen. When using the SCS curve number equation, the CPM compensates for this phenomenon by adjusting the curve number according to the frozen soil infiltration method of Holtan et al. (1975). If the average daily air temperature remains below 30°F for two weeks, hydraulic conductivity is decreased by 0.1 inch per hour for each degree below 30°F.

The SCS Curve Number equation is not directly sensitive to hydraulic conductivity. For a shallow soil with Antecedent Moisture Condition III, (SCS, 1964) the infiltration rate approximates the saturated hydraulic conductivity. Saturated hydraulic conductivities for each soil group are given in Table B-1. By regressing the hydraulic conductivity with the curve number for each group, the following relationship could be calculated:

$$\text{CN} = 99.55 - 10.72 \cdot k \qquad (r^2 = 95\%) \qquad (\text{B.29})$$

where CN = curve number

k = hydraulic conductivity, cm/hr

Then by decreasing hydraulic conductivity by 0.45 cm/hr for each degree (°C) of soil temperature below 0°C, the Curve Number for temperatures below freezing becomes:

$$\text{CN}_{fr} = \text{CN}_{orig} - 4.8 \cdot \text{SOILTEMP} \qquad (\text{B.30})$$

where CN_{fr} = curve number when soil is frozen

CN_{orig} = curve number for thawed soil, Antecedent Moisture Condition III

SOILTEMP = soil temperature, °C

This equation uses daily soil temperature, not the 14 day average air temperature. It was found in Section 7 that in New York, the 20-day average temperature was about 1.5°C lower than the temperature of a nearly frozen soil. Therefore, a decrease in conductivity is assumed to occur as soon as the soil temperature dips below 0°C.

Evapotranspiration--

The evapotranspiration model is based on the frequently used assumption that evapotranspiration will take place at the "potential rate" if the soil is near field capacity. The acutal evapotranspiration will be less than potential as the soil dries, and falls to zero as the soil approaches wilting point.

Potential evapotranspiration is computed by the sinusoidal function

$$Ep = \frac{Epmax}{2} \cdot (1 + \sin (\frac{JULDATE - 100}{182.5}) \cdot \pi) + Epbase \qquad (B.31)$$

where Ep is potential evaporation, cm/day

Epmax is the maximum yearly potential evaporation, cm/day

JULDATE is the Julian date

Epbase is the minimum yearly potential evaporation, cm/day

In winter, evaporation decreases with the square root of time after Zone 1 has reached field capacity. If the Zone 1 moisture content is less than field capacity, evapotranspiration is computed as a function of moisture content.

In the growing season, the plants use water from Zones 1 and 2 for their transpiration. Assuming that evapotranspiration decreases linearly with moisture content from field capacity to wilting point, the water loss due to evaporation from Zones 1 and 2 can be expressed as

$$ET(1)_{\Delta t} = SW(1)_{t-\Delta t} \cdot \exp \frac{-\Delta t \cdot Ep}{[DEPTH(1) + DEPTH(2)][FC - WP]} \qquad (B.32)$$

$$ET(2)_{\Delta t} = SW(2)_{t-\Delta t} \cdot \exp \frac{-\Delta t \cdot Ep}{[DEPTH(1) + DEPTH(2)][FC - WP]} \qquad (B.33)$$

where $ET(J)_{\Delta t}$ = evapotranspiration from Zone J in the period day t-Δt through day t, cm

Δt = number of days elapsed since the last rain or snowmelt

$SW(J)_{t-\Delta t}$ = soil water content of Zone J on day t-Δt, i.e. the day of the last event, cm

DEPTH(J) = width of Zone (J)

FC = moisture content at field capacity, cm^3/cm^3

WP = moisture content at wilting point, cm^3/cm^3

Sediment Sub-model--

The sediment sub-model, SED, uses the Universal Soil Loss Equation with cover factors from the new list composed by Wischmeier (1978) and an energy factor computed by either the Onstad and Foster (1975), Williams and Berndt (1977), or the Wischmeier and Smith (1965) method. The sediment models are described in Appendix C. The method to compute peak runoff for determination of energy factors is described in Appendix A.

Pesticide Sub-model

The pesticide sub-model consists of three parts: degradation and volatilization, downward displacement of pesticide bands, and pesticide in overland flow.

The attenuation of pesticide through volatilization and degradation is based on a stepwise first order degradation rate, as proposed by Donigian et al. (1977),

$$PEST_t = PEST_{t-\Delta t} \cdot \exp\left[-\Delta t \cdot \frac{0.69}{t_{\frac{1}{2}}}\right] \qquad (B.34)$$

where $PEST_t$ = quantity of pesticide in the soil at time t, g/ha

Δt = number of days elapsed since the soil pesticide content was updated

$PEST_{t-\Delta t}$ = pesticide concentration in soil on day $(t-\Delta t)$, i.e. the day that pesticide concentration was last updated

$t_{\frac{1}{2}}$ = half-life for pesticide degradation, days. Two different values for $t_{\frac{1}{2}}$ were used: one for the time between application and first runoff event after application, the other during the remainder of the year

The downward displacement of pesticides is modeled with a simple model proposed by, among others, Gardner (1965) and Rao et al. (1976). Diffusion is not considered. Instead, the average displacement of the midpoint of the pesticide band is computed as:

$$DISP = \frac{FLOW}{FC \cdot RETARD} \qquad (B.35)$$

$$RETARD = 1 + \frac{\rho_s \cdot K}{FC} \qquad (B.36)$$

where DISP = displacement of the band, cm

FLOW = quantity of flow past the pesticide band, cm

RETARD = retardation factor (dimensionless)

ρ_s = specific weight of soil, g/cm^3

K = adsorption partition coefficient, cm^3/g

Pesticide in Overland Flow

The following model used to determine the loss of pesticides in runoff was developed by Steenhuis and Walter (1978). The key assumption is that the concentrations of pesticides in the percolating water, runoff, and soil water are equal in the thin layer of soil, extending 0.9 cm down from the soil surface, designated as the mixing zone.

For a rainstorm of high intensity, the rainfall period is divided into three segments by the program. In the initial segment, the rainfall intensity is lower than the soils infiltration capacity, and all rainwater infiltrates. Pesticides are transported from the mixing zone to the lower zones by percolating water. The quantity of pesticide in the mixing zone at the end of segment is

$$\mathrm{PEST1} = \mathrm{PEST0} \cdot \left[\exp \frac{-\mathrm{WIBP}}{\mathrm{HMIX(SAT} + \rho_s \cdot \mathrm{K}}\right] \qquad \text{(B.37)}$$

where PEST1 = amount of pesticide in the mixing layer at the start of ponding, g · ha^{-1}

PEST0 = amount of pesticide in mixing layer just prior to rainfall, g · ha^{-1}

WIBP = water infiltrated before ponding, cm

HMIX = mixing zone depth, cm

SAT = moisture content at saturation, cm^3/cm^3

WIBP is obtained by

$$\mathrm{WIBP} = 0.2 \cdot \left(\frac{1000}{\mathrm{CN}} - 10\right) - \mathrm{STmax} \qquad \text{(B.38)}$$

If the value for WIBP thus calculated is less than the actual rainfall, then the latter value is used.

In the second and third rain segments, rainfall intensity is greater than infiltration capacity. Thus, not all rainwater can infiltrate. Before runoff can occur, however, the surface detention storages have to fill up. The second segment lasts from the first ponding until runoff begins. Pesticides are diluted in the ponded water as well as carried downward in the percolating water. The quantity of pesticide in the mixing zone at time when runoff begins is

$$PEST2 = PEST1 \cdot \left[\frac{STmax + HMIX \cdot (SAT + \rho_s \cdot K)}{HMIX \cdot (SAT + \rho_s \cdot K)}\right]^{\frac{WIBP}{P - WIBP}} \qquad (B.39)$$

where PEST2 is the amount of pesticide in mixing zone when runoff began, g/ha

In the third segment, pesticides from the mixing zone are lost both in runoff and in percolating water. The pesticide concentration in the mixing zone at the time when runoff stops is

$$PEST3 = PEST2 \cdot \exp - \frac{P - WIBP + \frac{(\rho_w \cdot K \cdot SED) \cdot R}{1 - SED}}{STmax + HMIX \cdot (SAT + \rho_s \cdot K)} \qquad (B.40)$$

$$LOSSW = \frac{R \cdot [1 + SED \cdot (\rho_w \cdot K-1)] \cdot (PEST2 - PEST3)}{(1 + \rho_w \cdot K \cdot SED) \cdot [(P - WIBP) \cdot (1-SED) + R \cdot SED \cdot \rho_w \cdot K)]} \qquad (B.41)$$

$$LOSSS = \frac{\rho_w \cdot K \cdot SED \cdot LOSSW}{1 + \rho_w \cdot K \cdot SED} \qquad (B.42)$$

where PEST3 = the amount of pesticide in the mixing zone when runoff stops, g/ha

LOSSW = loss of pesticide in solution in runoff water, g/ha

LOSSS = loss of pesticide adsorbed on sediment, g/ha

SED = sediment concentration in runoff of water, g/g

ρ_w = specific weight of water, g/cm^3

Equations (B.37) through (B.42) were based on a linear adsorption isotherm To simulate a nonlinear adsorption isotherm, a stepwise changing adsorption partition coefficient was introduced in the program.

Each range of pesticide quantity in the mixing zone is associated with a particular adsorption partition coefficient. The choice of adsorption partition coefficient for equations(B.37) through (B.42) depends on the pesticide concentration in the mixing zone.

Statistics

The statistical program which was developed for this project summarizes the results produced by the modeling program. It provides average monthly values of the variables, together with standard deviation and the maximum and minimum monthly values recorded during the entire time period. Average annual values also are provided along with the standard deviation and minimum annual values.

A second program analyzes data produced by the runoff model. It calculates the total number of days per month on which rainfall, runoff and soil losses of varying magnitudes were produced, and provides a table showing the probability of occurrence of events of varying magnitude on a month-by-month basis.

Both programs can easily be adapted to deal with any of the variables contained within the modeling program.[1]

PARAMETER VALUES FOR VALIDATION

The parameter values used in the CPM model for validation on the P2 watershed, Watkinsville, Georgia, are given below.

Plot Characteristics

length flow path	200 meter
slope	1.75%
soil group	B
active root zone depth	see Figure 7-4
crop	corn
area	1.29 ha

Hydrology

saturated hydraulic conductivity	0.5 cm/hour
suction at wetting front	22.5 cm

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[1] Developed by H. D. Murray-Rust. Research Assistant, Department of Agricultural Engineering, Cornell University.

surface storage depression	0.4 cm
interception by plants	0.2 cm
moisture content at saturation	0.39 cm^3/cm^3
field capacity	0.20 cm^3/cm^3
wilting point	0.14 cm^3/cm^3
maximum potential evapotranspiration	0.7 cm/day
minimum potential evapotranspiration	0.2 cm/day

Soil Loss

Universal Soil Loss Equation Parameters

K	0.28
P	0.5
LS	0.44

C Factor

Year 1973		1974		1975	
Date	C	Date	C	Date	C
4-18-73	0.40	4-23-74	0.20	5-11-75	0.15
5-11-73	0.62	5-29-74	0.18	6-19-75	0.13
6-09-73	0.54	8-11-74	0.14	8-09-75	0.11
7-30-73	0.42	9-14-74	0.28	8-29-75	0.10
8-30-73	0.24			10-04-75	0.19
11-02-73	0.33				

Pesticide Loss

Application dates and rates see Table 7-1

Atrazine

half-life after application	6 days
half-life after first runoff following application	16 days

Adsorption Partition Coefficient cm^3/g	Atrazine Quantity in Mixing Zone g/ha
1.5	above 400
5	150-400
20	91-150
30	78-91
100	50-78
1000	20-50
10000	10-20
100000	below 10

Paraquat
half-life 360 days

Adsorption Partition Coefficient cm^3/g	Paraquat Quantity in Mixing Zone g/ha
100	above 700
10000	10-700
100000	below 10

SYMBOLS USED

ADJAIRTEMP	adjusted air temp, °C
AIRTEMP	actual air temp, °C
CN	curve number
DISP	displacement of pesticide band, cm
DEPIMP	depth of the impermeable layer, cm
DEPTH(J)	width of layer J
Ep	potential evaporation, cm
Epbase	minimum yearly potential evaporation, cm/d

Epmax	maximum yearly potential evaporation
ET	evapotranspiration, cm
f	fillable pore space
FC	field capacity, cm
FLOW	quantity of flow past the pesticide band, cm
FLOWIJ	water flow from Zone I to Zone J, cm
HMIX	mixing zone depth, cm
hw	suction head at wetting front, cm
I	cumulative infiltration, cm
i	infiltration rate into the soil, cm/hr
i_b	infiltration rate in beginning period with constant intensity, cm/hr
i^*_{end}	infiltration rate at end of period if all water would have been infiltrated, cm/hr
INTCEP	interception of rain water by plants, cm
ST_{po}	surface storage at time of ponding, cm
SW(J)	soil water content of Zone J, cm
t	time, days
t_{per}	duration of rain period of equal intensity, hr
t_{po}	time when water starts ponding, hr
$t_{\frac{1}{2}}$	half-life, day
WATEREQ	water equivalent of snow, cm
WIBP	water infiltrated before ponding, cm
WP	moisture content at wilting point, cm
z	depth wetting front, cm
z*	hypothetical depth of wetting front if all water would have been infiltrated, cm

z_b	depth of wetting front at beginning of period with constant rain intensity, cm
z_{end}	depth of wetting front at end of period with constant rain intensity, cm
z^*_{end}	depth of wetting front at end of period with constant rain intensity if all rain water would have been infiltrated, cm
z_{po}	depth of wetting front when ponding starts, cm
ρ_s	specific weight of soil, g/cm^3
ρ_w	specific weight of water, g/cm^3
JULDATE	julian date
K	adsorption partition coefficient, cm^3/g
k	hydraulic conductivity, cm/hr
LOSSS	loss of pesticide adsorbed on sediment, g/ha
LOSSW	pesticide loss in dissolved phase, g/ha
M	snow melt, cm
MAXFLRATE	maximum flow out of the root zone, cm
P	rainfall, cm
p	rainfall intensity, cm/hr
p_{eff}	effective rainfall intensity, cm/hr
PEST	pesticide quantity in soil, g/ha
PEST0	amount of pesticide in mixing layer just prior to rainfall, g/ha
PEST1	amount of pesticide in mixing layer just prior to ponding, g/ha
PEST2	amount of pesticide in mixing layer just prior to runoff, g/ha
PEST3	amount of pesticide in mixing layer at time runoff stops, g/ha
R	runoff, cm

RETARD	retardation factor, dimensionless
SAT	saturated moisture content, cm^3/cm^3
SED	sediment concentration in runoff water, g/g
SOILTEMP	soil temperature, °C
ST	water stored in depressions, cm
ST_b	depression storage at beginning of period, cm
STend	depression storage at end of period with constant rain intensity, cm
STmax	maximum depression storage, cm

APPENDIX B REFERENCES

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APPENDIX C

SOIL LOSS PREDICTION MODELS

The primary function of SWCPs in humid regions is control of soil erosion. Techniques for predicting erosion have been primarily empirical in the past and produce the single output of quantity of erosion in units of tons/ac or MT/ha. The character of the eroded soil, its original location, and the distance it moved were not typically of concern. One reason for this approach was that conservationists were primarily interested in the soil resource remaining after erosion took place.

As the interest in sediment-related problems increased, attempts were made to predict sediment yield. The thrust of these attempts has been to piggy-back some additional functional relationships to account for distance of transport onto the established erosion prediction techniques. Obviously, the sediment predicting techniques derived this way have all the limitations inherent to the erosion models.

Perhaps the most serious limitation in the sediment models is that the character of the eroded soil is important but not included in them. Sediment character is important first, because its transport properties are dependent on the sediment particle size and density distributions; second, because the pollution problems caused by sediment differ depending on the sediment character; and third, because the amount of a substance adsorbed on the sediment depends on the adsorptive character of the particles it consists of as well as their original location in the field.

Since no model was available to deterministically estimate sediment yield and its characteristics, the approach taken in this project was to use the models that were available (empirical or partially empirical models) and to supplement the quantitative values they gave with qualitative information concerning sediment character. Sediment character depends on type of erosion, severity of erosion, type of storm, control mechanisms, etc.

For the transport of strongly adsorbed substances, two considerations of the sediment transport have to be made:

(a) the amount of sediment that is transported, and

(b) the preferential transport of fine clay and organic material as compared to coarser silt and sand particles.

Soil loss models are categorized as event-based or as average loss over some time period. Prediction of soil loss on an event basis is important if concentration of sediment or sediment-carried nutrient of pesticide is critical as a pollutant. Event-based models may also be used to predict long-term soil losses.

Average Annual Soil Loss Models

Early methods to predict soil erosion were developed for the purpose of soil conservation (i.e., to preserve long term soil productivity). A very extensive data base was collected to develop the most widely used equation for estimation of erosion from cropland, commonly known as the Universal Soil Loss Equation (USLE). This equation is described in the popular publication (currently being revised), Agricultural Handbook No. 282 (Wischmeier and Smith, 1965).

Erosion-index distribution curves have been developed by Wischmeier and Smith (1965). They give the percent of annual erosion index for each month of the year. These values can be used directly to determine relative magnitude of the rainfall factor for various times of the year.

Event-Based Models

At the present time, various models are in use to estimate edge of field losses of sediment. Some models are still in conceptual form, others are incomplete, and/or require data not routinely available (Woolhiser and Todorovic, 1974; Meyer and Wischmeier, 1969). Three models for predicting event-based soil losses were used in this study. They include the USLE (Wischmeier and Smith, 1965), Onstad-Foster Model (Onstad and Foster, 1975), and the Williams Model (Williams, 1975). The primary difference between the three techniques is the way in which the storm event energy term is calculated.

The USLE equation is used on an event basis by replacing the annual rainfall erosion index factor (R) by the storm erosion index. The storm erosion index is the product of the rainfall energy and the maximum 30-minute rainfall intensity. The rainfall energy term is given in equation 7 of Appendix A.

Williams (1975) concluded that sediment loss was better correlated with runoff than with rainfall. He, therefore, tried a number of combinations of peak flow rate and total runoff for the rainfall factor in USLE. The runoff parameter which gave the best fit between predicted and measured sediment yield on 18 watersheds was:

$$W = 20.91A^{0.12} (Q \times qp)^{0.56} \tag{1}$$

where Q and qp are total and peak runoff in units of cm and cm/hr, respectively, and A is watershed area in hectares. When the right side of equation 1 is substituted into the USLE, the predicted sediment is in metric tons.

Onstad and Foster (1975) presented an erosion-deposition model based on a modified form of the USLE incorporating hydrologic variables of runoff and rainfall in the energy term. This model is discussed in Appendix A.

Soil losses for the 72 storm events which occurred in the years 1973 to 1975 on Plot P2 in Watkinsville, Georgia, were predicted with the event-based USLE, the Onstad-Foster model and the Williams model using observed runoff and peak runoff values. Watershed parameters are given in Appendix A. The observed and predicted values (ordered by order of magnitude of the observed values) are given in Table C-1. Rainfall records are available or can be extrapolated to almost any watershed. Runoff records, however, are not available for most watersheds so total and peak runoff generally need to be calculated. Techniques used to predict total and peak runoff are given in Appendix A.

Runoff

Predicted values for total and peak runoff for single storm events were not as close to observed values as hoped for. Figures C-1 and C-2 are regressions of observed versus predicted total and peak runoff for 72 events on the P2 watershed in Watkinsville, Georgia. The regression coefficients (R^2) for total and peak runoff were 55 and 50 percent, respectively. For the 72 storms which were evaluated a total of 43.8 cm of runoff was observed while 38.5 was predicted.

If the calculated values of total or peak runoff for a storm were zero, the energy term of the Williams model was also zero, that of the USLE was unaffected and the Onstad-Foster energy term was half that of the USLE.

Soil Loss

Since total and peak flow were known for the test watershed the soil losses predicted by the Onstad-Foster and Williams models were calculated directly from the field data. In general, runoff data are not available, so total and peak runoff were also calculated for comparative purposes when used in these two models.

As shown in Table C-1 all three models overpredicted total soil loss for the 72 storm events when actual runoff and rainfall data were used. The event-based USLE, Onstad-Foster model and Williams model overpredicted total soil losses by 133, 112, and 84 percent, respectively. Based on total soil losses for all storm events the Williams model would appear to give the best predictions. However, if only the largest storms are considered, the reverse is true. For the ten largest recorded soil loss events the event-based USLE gave better total soil results than the Onstad-Foster model or the Williams model. Even for the ten largest events, all models still overpredict soil losses. For very small storms when little or no runoff was observed the Williams model, as expected, gave the best results.

TABLE C-1. SOIL LOSS AS OBSERVED AND PREDICTED WITH 3 DIFFERENT MODELS FOR WATERSHED P2 IN WATKINSVILLE, GEORGIA (1973-1975). (LOSSES IN KG/HA)

Date	Observed	USLE	Actual Q and qp Onstad-Foster	Actual Q and qp Williams	Calculated Q and qp Onstad-Foster	Calculated Q and qp Williams
1/8/75	0	20	10	0	10	0
9/17/75	0	290	150	60	170	120
9/12/75	0	40	20	0	20	0
6/8/73	0	110	50	0	60	70
8/27/75	0	280	140	0	140	0
5/3/75	0	400	200	0	250	280
9/6/75	0	20	10	0	10	0
3/31/75	0	170	80	0	100	80
9/30/73	0	90	40	0	40	0
11/21/73	0	230	110	0	110	0
11/28/73	0	50	20	0	20	0
2/17/75	0	240	130	30	210	300
12/15/73	0	90	40	0	70	90
3/12/75	0	210	100	0	170	210
12/30/73	0	110	60	10	100	100
3/24/75	0	440	220	0	230	100
1/20/74	0	80	40	0	40	0
2/6/74	0	330	190	70	450	420
2/14/74	0	40	20	0	20	0
2/15/74	0	80	50	10	40	0
2/22/74	0	50	30	20	20	0
3/21/74	0	230	110	0	110	0
12/19/74	0	90	50	80	80	80
12/15/74	0	220	120	10	240	260
3/14/75	0	160	80	0	200	280
1/12/75	0	150	80	20	140	170
5/12/74	0	90	40	0	40	0
1/24/75	0	250	140	20	230	280
8/7/74	0	120	60	0	60	0

TABLE C-1. SOIL LOSS AS OBSERVED AND PREDICTED WITH 3 DIFFERENT MODELS FOR WATERSHED P2 IN WATKINSVILLE, GEORGIA (1973-1975). CONTINUED.

Date	Observed	USLE	Actual Q and qp Onstad-Foster	Actual Q and qp Williams	Calculated Q and qp Onstad-Foster	Calculated Q and qp Williams
7/26/74	0	80	40	0	40	0
12/29/74	0	140	80	10	130	160
11/20/74	0	180	90	0	90	0
1/19/75	0	70	30	0	50	70
12/24/74	0	90	40	0	80	50
3/29/74	1	340	180	30	170	0
12/21/73	1	190	40	20	150	210
3/16/75	1	80	50	20	60	70
12/7/73	2	600	310	30	390	310
2/4/75	2	260	140	30	310	390
7/30/73	2	950	500	100	470	0
8/29/74	3	80	50	40	40	0
2/24/75	3	360	190	40	320	370
1/10/75	3	370	190	40	300	350
9/23/75	5	70	40	20	30	10
2/18/75	6	170	220	380	140	190
8/17/74	7	90	50	40	50	0
4/4/74	9	470	310	260	270	180
5/5/74	10	110	60	20	50	0
4/13/74	14	670	420	270	340	60
8/10/74	22	490	260	90	240	0
4/2/75	22	1810	1480	1250	1900	1640
9/8/73	23	1790	960	230	950	370
7/24/74	23	140	80	80	70	0
5/16/75	30	120	110	70	150	310
9/13/73	33	100	240	630	50	0
5/14/75	40	250	140	110	120	0
12/31/73	54	970	770	630	1550	1420
3/13/75	58	2430	2460	1650	2910	2140

TABLE C-1. SOIL LOSS AS OBSERVED AND PREDICTED WITH 3 DIFFERENT MODELS FOR WATERSHED P2 IN WATKINSVILLE, GEORGIA (1973-1975). CONTINUED.

Date	Observed	USLE	Actual Q and qp Onstad-Foster	Actual Q and qp Williams	Calculated Q and qp Onstad-Foster	Calculated Q and qp Williams
8/16/74	70	830	530	330	450	200
5/23/74	92	1170	710	350	1020	1060
5/7/75	110	750	450	290	510	560
5/31/75	215	400	260	210	240	240
7/13/75	320	460	370	520	230	0
7/8/73	422	2990	2720	2770	1540	400
7/24/75	427	380	590	870	200	80
6/13/73	530	470	680	1540	240	0
7/27/74	661	4340	3380	2180	2410	940
5/23/73	716	350	560	1300	180	0
6/27/74	966	3220	2780	2240	1880	1030
6/6/73	1108	2180	3020	4580	1090	0
6/11/75	4122	1440	1620	1700	1080	960
5/28/73	8240	5590	9010	8420	6700	7780
TOTAL	18373	42800	38600	33720	32410	24390
TOTAL (10 largest observed events)	17512	21420	25730	30800	17620	11190

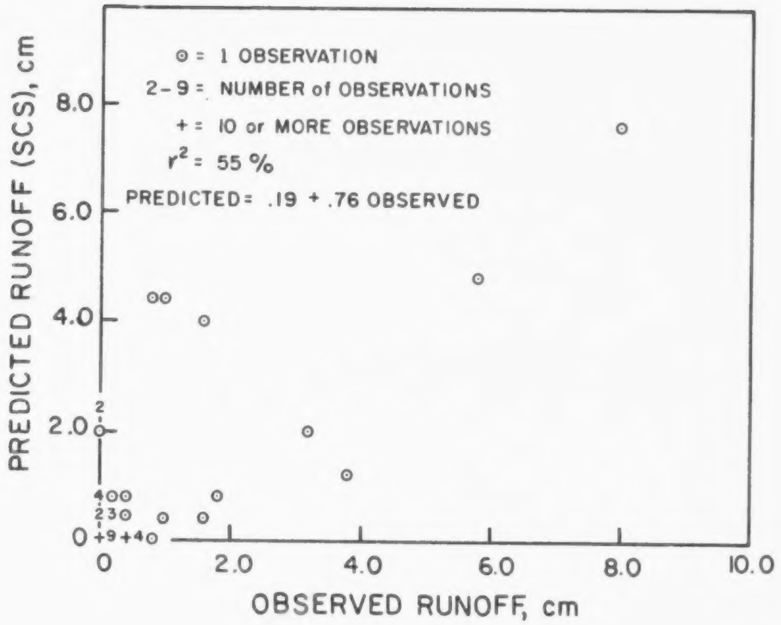

FIGURE C-1. PREDICTED RUNOFF (SCS) VS. OBSERVED RUNOFF.

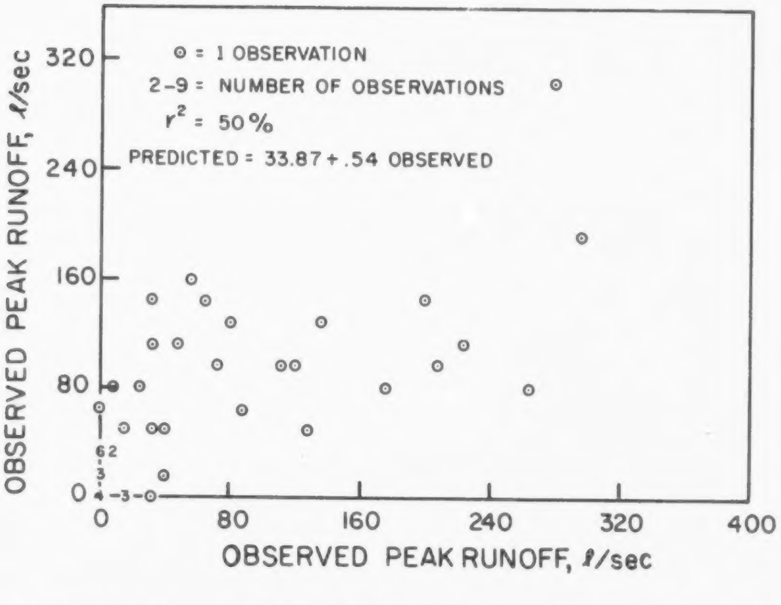

FIGURE C-2. PREDICTED PEAK RUNOFF VS. OBSERVED PEAK RUNOFF.

The three models rarely underpredicted soil loss and when they did it was only for large soil loss events (i.e., greater than 215 kg/ha soil loss). The event-based USLE underpredicted soil losses on five storm events, while soil loss was underpredicted only twice by each of the other two models. The largest difference between observed and predicted results for soil loss seem to be for a few events, when the models grossly over-estimate soil losses. For example, on 8 July 1973, 422 kg/ha of sediment was recorded leaving plot P2 while all three models were in close agreement in estimating soil losses at nearly 3000 kg/ha.

For the 25-year model simulation referred to in Section 5 only rainfall data were available so total and peak runoff had to be calculated. Soil loss predictions for the Onstad-Foster model and the Williams model using calculated total and peak flow are also given in Table C-1 for the storms of 1973 to 1975. Regression analysis showed that the correlation between observed and predicted soil loss was not much changed when actual instead of predicted runoff data were used (r^2 = 71 and 72% for Williams' model and 71 and 68% for Onstad-Foster model when observed and predicted runoff data were used respectively). Total and particularly peak runoffs were underpredicted for large storm events (see Figures C-1 and C-2). This probably compensated for the occasional large over estimation of soil losses when actual runoff data were used. This point is also illustrated by the soil loss predicted with the Williams model for the ten largest observed soil-loss events. The Williams model very much under estimated soil losses for these selected large events due to the fact that no runoff was predicted.

Summary and Conclusions

The Williams model is a function only of runoff, the event-based USLE depends only on rainfall, and the Onstad-Foster relies on both. Runoff data are typically not available so they must be calculated from rainfall data and watershed physical characteristics. All three models over-estimated total soil losses for a three-year period, with the Williams model predicting closest to observed values. The Williams model worked very well for small events but was the poorest of the three models for the ten events with highest observed soil losses. No model was found to be clearly more accurate than the others in all cases. Furthermore, since the comparison was made on only one watershed for a three-year period, extrapolation to other watersheds needs to be handled with care. Theoretically, inclusion of runoff parameters in a sediment estimating model seems to be appropriate and advisable in future model development.

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APPENDIX D

CONTROL OF NITROGEN LOSSES BY MANAGEMENT OF FERTILIZER APPLICATIONS

While SWCPs must sometimes be analyzed separately in order to quantify their effectiveness in reducing potential water pollution, they are in reality used in concert with a total farming system. Total nutrient loss is best controlled with a combination of control of soil loss and good management of fertilizer (McDowell and Grissinger, 1977).

Control of agricultural chemicals is much more complex once they are applied to the land than before. Therefore, careful management of chemical applications is advisable. If fertilizer is applied close to the time of crop uptake and does not exceed the crop requirement, potential nutrient losses will be minimized. The most effective application time and technique may not be the least expensive. Usually, the time and method of fertilization depend on seasonal fertilizer cost, availability of application equipment and of farm labor (Whitaker et al., 1978). For example, fall application of nitrogen fertilizer is sometimes used to spread the work load throughout the year. Some farmers prefer to apply more chemical than recommended as a form of "insurance" that enough will be available.

The opportunities available for source management of applied nitrogen include application rate and techniques, timing, choice of field and form of nitrogen fertilizer. When manurial nitrogen is used one can choose application rates and field but in many situations the timing and technique of manure disposal is dictated by the manure handling system.

The obvious source management technique to control nitrogen losses is reduction in rates of application. Chichester (1977) investigated nitrogen losses from two rates of applications on corn in Coshocton, Ohio. His results for inorganic nitrogen losses in percolation and surface runoff are shown in Table D-1. The lysimeters contained a moderately well-drained Keene silt loam soil on a six percent slope. Inorganic nitrogen losses in surface runoff in this experiment were relatively small while percolate losses were very large. Nitrogen application rates used by Chichester were apparently so high that nitrogen built up in the soil profile. This is indicated by the comparatively high nitrogen losses even in 1973 when no nitrogen was applied. These results illustrate that excessive nitrogen application rates can result in large nitrogen losses to percolation.

TABLE D-1. INORGANIC NITROGEN LOSSES FROM TWO SYSTEMS OF AMMONIUM APPLICATION TO CORN (ADAPTED FROM Chichester, 1977)

Lysimeter	Application Date	Application Rate (kg/ha)	Period of Record	Inorganic N Percolate (kg/ha)	Inorganic N Runoff (kg/ha)
A	Apr 1971	336	1971-72	179	1
	May 1972	336	1972-73	244	6
	-	0	1973-74	133	3
	May 1974	178	1974-75	84	7
	TOTALS	850		640	17
B	Apr 1971	672	1971-72	202	3
	May 1972	336	1972-73	274	10
	-	0	1973-74	165	6
	May 1974	178	1974-75	115	11
	TOTALS	1186		756	28

Whitaker et al. (1978) reported that for shallow Missouri soils, nitrogen in runoff increased as rate of application was increased from 15 to 368 kg/ha. Results from this study are shown in Table D-2. This study was conducted on plots 3.2 x 27.4 meters on a three percent slope. The soil was underlaid by a claypan 18 to 45 cm below the soil surface, which restricted water movement. Metal barriers were placed around the plots down to the claypan to prevent subsurface flow out of the plots. All flow off the plots was collected at the bottom of each plot. Losses of nitrogen reported by Whitaker et al. (1978) were much lower than those reported by Chichester (1977) even at the highest application rate. The difference could be accounted for by greater denitrification or by losses through the claypan in the investigation by Whitaker et al. Maximum yields in the study by Whitaker et al. were achieved at nitrogen application rates of 198 kg/ha. Corn yields at application rates higher than 187 kg/ha did not increase but soluble nitrogen losses in the runoff did, indicating that fertilization in excess of crop requirements can result in increased nitrogen losses of a field.

The CNS model was used to evaluate nitrogen losses for continuous corn crop systems in three locations - New York, Georgia and Iowa - over 25 years. The physical features of the particular fields modeled are given in Table 6-7. The "base" management scheme was an application of 75 percent of the total nitrogen requirements of the corn crop as given in Table 6-7 applied in the form of nitrate at plant emergence. Mineralization of soil organic nitrogen was assumed to be 15 to 45 percent of the crop nitrogen needs. The base management scheme is not necessarily the recommended management system for any of the locations. Crop yields were assumed to remain constant for all 25 years modeled.

TABLE D-2. SOLUBLE NITROGEN LOSSES FROM CORN PLOTS (Adapted from Whitaker 1978)

Application Rate (kg/ha)	Losses in Runoff[a] (kg/ha)
15	7.9
98	14.7
198	14.9
214	17.0
252	39.6
368	35.1

[a] Three-year average

Table D-3 shows nitrogen losses for different application rates as compared to the base condition. Inorganic nitrogen losses for the base condition in June were 36, 31, and 29 kg/ha for New York, Georgia and Iowa, respectively. If the fertilizer rates were twice those of the base condition, the expected losses would increase 86 to 93 percent. On the other hand, for the New York case at least, if the base rate of fertilizer was halved, the expected losses would be reduced from 36 to 26 kg/ha.

A second technique for good management of nitrogen fertilizer is application at the time the plant most needs it. Ideally this would require a split application. Whitaker et al. (1978) recommend a split application with a small amount at planting time and the rest 5 or 6 weeks later. The base conditions of Table D-3 do not include this ideal approach, but timewise it is the best of the alternatives given in the table. Fall applications typically cause losses 100 to 148 percent greater than the base condition. The values in Table D-3 were based on nitrate fertilizers so that the nitogen was available to leach out of the soil immediately after application. The losses of inorganic nitrogen in preplant applications are less than those for fall but are still substantial. The model indicated that fall applications of nitrate could result in annual nitrogen losses which exceeded the application rate when the base rate was used. In Iowa this practice resulted in nitrogen losses of about 70 percent of the amount applied.

The expected losses of nitrogen increase relatively more in Iowa than in New York or Georgia when twice the base rates are fall applied. However, a lower percent of the applied nitrogen is lost in Iowa than in the other two locations for the fall base application.

As shown in Chapter 6, surface runoff with conventional tillage was typically 20 to 25 percent of the total of surface runoff and percolation. Dissolved nitrogen is lost in surface runoff and leachate in about the same proportions as the water movement. Total inorganic nitrogen losses were greater in the leachate than in the surface runoff, but yearly variations over the 25-year simulated period were proportionately much higher in the surface runoff. For example, nitrogen applied in April in New York at the base rate resulted in a mean and standard deviation for leachate losses of 43 and 4 kg/ha while for the surface runoff the mean and standard deviation were 7 and 3 kg/ha, respectively.

TABLE D-3. NITROGEN LOSSES FOR SELECTED APPLICATION RATES AND TIMES WITH CNS MODEL. (kg/ha)

	Application Date	Application Rate (kg/ha)		
New York		142	71 (Base)	36
	Sept	122	73	
	Mar	97	57	39
	Apr	87	51	34
	June(Emergence)	67	36	26
		Application Rate (kg/ha)		
Georgia		116	58 (Base)	
	Sept	104	62	
	Mar	76	47	
	Apr	67	39	
	May(Emergence)	58	31	
		Application Rate (kg/ha)		
		202	101 (Base)	
	Sept	126	72	
	Mar	95	52	
	Apr	70	41	
	May(Emergence)	56	29	

Table D-4 illustrates the expected time of nitrogen losses at the three locations when nitrogen is applied at the base rate. The 25-year simulation period indicated that when nitrate was applied at plant emergence, losses in New York would be fairly uniformly distributed in all seasons, while in Iowa and Georgia, losses could be expected to be greatest in the spring and smallest in the fall. When nitrogen was applied in March, several months before the crop could us it, very high losses occurred in all three locations immediately following application. September application of nitrate resulted in high losses in fall and winter in New York and Georgia, while for this situation, losses in Iowa were particularly high in spring.

Since nitrogen was assumed to be applied in nitrate form it was immediately available for crop uptake or transport with water. The temporal variations in nitrogen losses listed in Table D-4 are a result of seasonal water movement, crop uptake, and application timing. If other nitrate transformations, particularly denitrification, were included in the model the distributions of losses might be different. Smid and Beauchamp (1976) have shown that at typical application rates all nitrate can denitrify in only one or two days if conditions are favorable for denitrification. Conditions favorable for denitrification are fairly common on some cropland. If this were the case (e.g., denitrification was important) total nitrogen losses would be greater but losses to runoff and leachate would be smaller.

TABLE D-4. AVERAGE SEASONAL NITROGEN LOSSES IN KG/HA WHEN NITRATE IS APPLIED AT THE BASE RATE AS PREDICTED BY THE CNS MODEL FOR A 2-YEAR PERIOD

	Application Time								
	May			March			Sept		
Season[a]	NY[b]	Iowa	Ga	NY	Iowa	Ga	NY	Iowa	Ga
Winter	11	4	11	8	3	11	25	15	25
Spring	9	16	11	38	42	32	15	34	7
Summer	8	9	6	5	4	1	1	3	0
Fall	8	0	0	3	1	1	30	20	38

[a] Winter, Dec-Feb; Spring, Mar-May; Summer, June-Aug; Fall, Sept-Nov.

[b] Application date June 1.

Summary

The CNS model was used to simulate 25-years of nitrogen losses at three locations for selected nitrate application rates and times. Results indicated that for all locations, losses increased substantially as the time period prior to planting was increased. Average expected nitrogen losses approach or exceed the quantity applied for fall application. When the nitrate application rate was double the quantity taken up by the crop, losses typically increased by about 70 percent over those at the base rate.

REFERENCES APPENDIX D

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APPENDIX E

CALCULATION OF BUDGETS, SOIL EROSION, AND SEDIMENT DELIVERY FOR THE LINEAR PROGRAMMING MODEL

Budgets and Constraints

A separate Linear Programming (LP) model was used for each of the example farms (New York, Iowa, and Texas). Each model is similar to the others. Therefore, to avoid repetition, the New York example farm will be discussed, followed by a discussion of the similarities and differences of the Iowa and Texas farms.

The New York example farm is a dairy farm, where the primary source of income is milk sales. The maximum number of cows is 80, with an average replacement rate of 25%. To feed the dairy herd and heifers, three least cost feed rations using 1978 feed prices were used (Knoblauch et al. 1978). They are: (1) hay crop base, (2) one-half corn silage base and (3) corn silage base. Both forages (hay and silage) must be grown on the farm. Corn grain may be grown or bought as needed in the ration, and soybean meal can only be bought. These three alternatives allow the least cost ration to be selected from an almost total row crop operation (corn silage base) down to a total sod crop operation (hay crop base) if necessary. The Iowa example farm is a cash crop/hog feeding operation with a nutrient requirement ration. The Texas farm is cash crop and hay.

The amount of land in New York is limited to 70 hectares which must grow all the corn silage and hay used by the dairy herd. Of this, one-half is moderately well drained, highly productive soil and has an 8% slope (Honeoye-Lima). The remaining one-half is somewhat-poorly drained soil with a 4% slope and has lower yields (Lima-Kendaia). The Iowa farm is 100 hectares of Muscatine-Tama-Garwin Association which consists of the following soils: 40% Tama, 30% Muscatine, 15% Dinsdale, and 15% Garwin. The Texas farm is 100 hectares of Houston Black-Heiden soil Association, consisting of three soils: 45% Houston Black Clay, 35% Heiden Clay, and 20% Trinity Clay.

Crop budgets for each soil type used in Iowa were provided by Jim McGrann (Personal Communication, 1978) and budgets for the Texas Blackland Region were provided by Cecil Parker (Personal Communication, 1978).

In New York, dairy farmers grow three basic feeds: corn grain, corn silage, and hay. These crops can be grown using conservation tillage or no-till, straight-row or contouring, and continuously or in rotation. Two rotations are considered, four years corn and four years hay or one year

TABLE E-1.

CROPPING COMBINATIONS

New York Cropping Combinations	
Corn silage chisel	On contour
Corn silage no-till	Diversion system # 1***
Corn silage chisel 4 x 4 rotation**	Diversion system # 2
Corn silage chisel 1 x 4 rotation**	Terrace system # 1***
Corn silage no-till 4 x 4 rotation	Terrace system # 2
Corn silage no-till 1 x 4 rotation	Terrace system # 3
Corn grain chisel	Terrace system # 4
Corn grain no-till	
Corn grain chisel 4 x 4 rotation	
Corn grain chisel 1 x 4 rotation	
Corn grain no-till 4 x 4 rotation	
Corn grain no-till 1 x 4 rotation	
Iowa Cropping Combinations	
Corn grain	On contour
Corn grain, soybeans	Terrace system # 1
Corn oats, meadow, meadow, meadow	Terrace system # 2
Corn, corn, oats, meadow, meadow	Terrace system # 3
Corn, corn, corn, oats, meadow	Terrace system # 4
Texas Cropping Combinations	
Cotton	On contour
Wheat	Terrace system # 1
Sorghum	Terrace system # 2
Hay	Terrace system # 3
Cotton, cotton, sorghum	Terrace system # 4
Cotton, sorghum, sorghum	
Cotton, wheat, wheat	
Sorghum, wheat, wheat	
Cotton, sorghum	
Cotton, sorghum, wheat	

* All possible combinations of the two columns were used.

** 4x4 Rotation = four year corn and four year hay
1x4 Rotation = one year corn and four year hay

*** Terrace and diversion systems are explained later in this Appendix.

TABLE E-2.

NEW YORK BASE BUDGET-CONSERVATION TILLAGE

	Corn Grain	Corn Silage	Haylage
Machinery Cost			
Tillage and planting	$ 10.19	$ 10.19	$ 3.27
Cultivation	1.88	1.88	--
Harvest[&]	8.15	16.68	47.44
Fertilizer			
Nitrogen	37.80	37.80	--
Phosphorus and potassium	21.10	21.10	23.50
Pesticide and Herbicide	31.85	31.85	18.50
Miscellaneous	38.74	34.72	37.07
Sub Total	$149.71	$154.22	$129.78
Labor (hr.)			
Spring	(2.3) 8.60	(2.3) 8.60	(0.9) 3.40
Summer	(0.7) 2.60	(0.7) 2.60	(17.2) 64.50
Fall	(4.7) 17.60	(10.5) 39.40	(0.2) 7.50
Total	$178.51	$204.82	$205.18
Yield MT/ha[&&]	5.19	36	17.8

* Cost per hectare includes only variable cost

& Harvest costs vary according to yield

¢ Fertilizer use varies according to soil response

&& Yields vary according to soil type and response to rotation corn and four years hay. The cropping practices shown in the first column of Table E-1. can be used in conjunction with the terrace or diversion systems shown in the second column to make 84 possible combinations of SWCPs for each soil type in New York. Likewise there are 25 for each soil type in Iowa, and 50 for each soil type in Texas (Table E-1). Some of these practices, however, may not be feasible, for example, working on the contour is only effective for shorter slope lengths.

For New York, budgets for planting, cultivating and harvesting operations (using conservation tillage) were constructed (Table E-2). Machine time efficiency, repair cost and fuel consumption information came from "Selecting Field Machinery" Campbell (1973). The machinery complement which goes with a nine foot chisel plow and four row planter were used. In New York, neither conservation tillage nor no-till have a significant effect on yields for the soils on the example farms. (Fred Swader, Personal Communication, 1978). In Iowa this is true for conservation tillage, however, yield decreases of around 5% are expected with no-till (Minora Ameniya, Personal Communication, 1978). In Texas, yield decreases are expected with both reduced tillage practices, thus neither is widely used on the soils of the example farm.

Some no-till budgets were also used in New York because the yield response is the same as for conservation tillage (Fred Swader, Personal Communication, 1978). The no-till budgets show a decrease in machinery and labor expense but an increase in herbicide and pesticide expense for a net increase of $4.00 per hectare (Table E-3). In Iowa a reduction in yield is expected with no-till (William Shrader, Personal Communication, 1978), making it a poorly accepted practice. Thus it was not budgeted for Iowa.

TABLE E-3.

PARTIAL BUDGET - NEW YORK NO-TILL OPERATION VS. CONSERVATION TILLAGE

	Cost of Conservation Tillage	Cost of No-till
Spring Machine	$10.19	$ 5.99
Spring Labor	8.60	4.55
Summer Machine	1.80	1.14
Summer Labor	2.60	1.09
Herbicide and Pesticides	31.85	46.35
	$55.12	$59.12

Terracing, diversion ditches, and contouring require more turning time and an extra trip around the field to work the end rows and corners. Hence, machine times were increased by 10% and labor increased by 12%.

Installation cost for terraces with tile drainage is \$6.50 per meter and diversions cost \$2.50 per meter in New York (Lauren Johnson, Personal Communication, 1978). Maintenance costs for terracing and diversions ranged from 1.5% to 3.5% of installation cost with the higher percentage going to structures with wider intervals. For both terraces and diversions, construction cost are amortized over an expected life of 40 years at an 8% interest rate. The annual cost per hectare for each terrace system was calculated and added to the cost of growing and harvesting the crop using conservation tillage. Because diversions take a percentage of land out of production, that percentage of the budgeted cost are subtracted from the base budget before adding the annual cost for diversions.

In Iowa the installation cost of tile drain terraces varied according to slope of land being terraced. On Muscatine and Garwin soils, the cost was \$7.35 per meter, Tama soil cost was \$7.85, and Dinsdale soil cost was \$8.40 (Jim McGrann, Personal Communication, 1978). In Texas the cost of terraces was \$0.46 per meter plus \$25 per hectare for construction of the grassed waterway used for drainage (Cliff Williams, Personal Communication, 1978).

New York crop yields remained the same on each soil type for all production practices except where diversion were used. In this instance, the amount of land used by the diversion was taken out of production and a poor quality low yield hay was used instead. On all soil types in Iowa, corn grain yield increased 5% when preceded by one, two or three years of meadow production. In Texas grain sorghum production increased by 23% when following cotton in the rotation, and wheat production decreased 5% after cotton or grain sorghum production.

In all three areas, each soil type results in different yields for each crop, giving some soil types a comparative advantage over other soil types for producing a particular crop. However, a different crop may gain the comparative advantage on a given soil type due to changes in production practices needed in order to meet soil loss standards. There are two reasons for this changing advantage: The first is that production costs for a different crop may be less then production cost of the original crop when meeting soil loss standards. The second reason is that the original crop may have to be grown in rotation with another crop in order to meet the soil loss standards.

For each SWCP used with each crop, soil and sediment losses per hectare were calculated (shown later). Using varying levels of soil erosion as the constraint, the LP was allowed to pick the most profitable combination of crops and production practices.

Soil Erosion Calculations

Soil erosion is calculated from the Universal Soil Loss Equation (USLE) (Wischmeier, *et al*. 1965). This equation is given by:

$A = 2.25\ R\ K\ L\ S\ C\ P$ where

A = Average Annual gross soil erosion in MT/ha.

2.25 = Metric conversion factor from tons/acre to metric tons per hectare.

R = Rainfall erosivity factor

K = Soil erodibility factor

L = Slope length factor

S = Slope gradient factor

C = Cropping practice factor

P = Conservation practice factor

R, K and S Factors

The values for R, K, and S are independent of SWCPs and are given in Table E-4.

TABLE E-4.

R, K, and S FACTORS FOR THE EXAMPLE FARMS

State and Soil Type	R	K	S**
New York			
Honeoye-Lima	100*	.32*	.840
Lima-Kendaia	100*	.32*	.350
Iowa			
Muscatine	175***	.28***	.117
Tama	175***	.28***	.304
Dinsdale	175***	.32***	.701
Garwin	175***	.28***	.117
Texas			
Houston-Black	320****	.32****	.182
Heidon	320****	.32****	.260
Trinity	320****	.32****	.089

* Swader (1974)

** $S = (0.43 G^2 + 0.30 G + 0.43) / 6.613$
where G = slope gradient in percent (Wischmeier, et al. 1965).

*** Earnest Hintz, Personal Communication, 1978.

**** Cliff Williams, Personal Communication, 1978.

L Factor

The L factor is a function of the slope length. On simple slopes (uniform slope) $L = \sqrt{\frac{\lambda}{22.73}}$ where λ is the slope length in meters, or the interval between structures where these have been implemented (Wischmeier, *et al*. 1965). On complex slopes, L is calculated separately for each segment of the slope having a different gradient. Thus for the example farms, when no terraces or diversions were installed, the slope length was the distance from the top of the farm to the stream, and for each soil type L was given by the formula:

$$L = (\lambda_j^{1.5} - \lambda_{j-1}^{1.5}) \Big/ \sqrt{22.13} \; (\lambda_j - \lambda_{j-1})$$

where λj = Slope length in meters from top of slope to bottom of soil type j, and λ_{j-1} = slope length from top of the slope to top of soil type j. When terraces or diversions are implemented, the slope length is calculated using the intervals, between structures for λ. These intervals are given in Table E-5. Note that Table E-5 lists all structures which were budgeted and entered into the LP program. Many of the structures did not actually appear in any of the farm plans.

TABLE E-5.

INTERVALS BETWEEN STRUCTURES FOR THE EXAMPLE FARMS

State and Soil	D_1*	D_2*	T_1*	T_2*	T_3*	T_4*	T_5*
New York							
Honeoye-Lima	242	75.3	242**	119**	56.5**&	122	58&
Lima-Kendaia	121	56.5	242**	119**	56.5**&	122	58&
Iowa							
Muscatine	---	---	67&	142	142	150	--
Tama	---	---	32&	92	192	192	--
Dinsdale	---	---	29.5&	67	67	75	--
Garwin	---	---	69&	67	75	75	--
Texas							
Houston-Black	---	---	31&	49	104	104	--
Heiden	---	---	24&	39	90	90	--
Trinity	---	---	50&	50	50	44	--

* D represents a diversion ditch system; T represents a terrace system. The number subscripts are keyed to specific intervals evaluated for the example farms.

** Terrace next to stream

& Meets SCS specifications (USDA-SCS, 1969).

C Factor

For the New York and Iowa example farms, C factors were derived from recently revised tables furnished by Walter Wischmeier (Personal Communication, 1978). For the Texas example farm, C factors are from a recent study in that area on the Lavon Reservoir Watershed (Taylor et al. 1978). The effectiveness of cropping practices for reducing erosion depends, among other things on crop yields. Thus C factors may vary from soil to soil on a farm depending on yield levels. This effect was taken into account for both the New York and Iowa farms, but not in Texas due to insufficient data. Table E-6 lists the C factors used for each rotation on the example farms.

P Factor

The P factor is 0.5 for all contouring, terraces, and diversions except where slope gradients are less than 2%. In this case 0.6 is used. The strip-cropping P factor is one half the contouring factor. Where neither contouring nor structures are used, the P factor is 1.0.

Sediment Delivery

Sediment delivery is calculated as the product of the level of soil erosion and the sediment delivery ratio. The method used for estimating the sediment delivery ratio is fairly arbitrary and based on a loose interpretation of data collected and analyzed by Renfro (1974). Thus it is not expected that the procedure used would necessarily give accurate estimates of the SDR in actual field situations. However, the results of the analysis are assumed to depict the relationship between and relative importance of different areas with differing SDR's when control of sediment loss is desired.

Calculation of the SDR is based on the formula $SDR = 2.5d^{-.36}$ where d = the distance from the center of a field to a waterway. Waterways include both natural bodies and man-made structures such as diversions and terrace channels. For fields directly bordering a waterway, the lower 125 meters of slope are assumed to have an SDR of 1.0. The distance formula is applied to calculate the SDR for the area above the lower 125 m and the SDR for the field is an average weighted according to the proportion of soil erosion occurring in each area. For areas above tile-outlet terraces, the calculated SDR is multiplied by 0.2 to account for redeposition in the terrace channel (Walter Wischmeier, Personal Communication, 1978). A factor of 0.4 is used for terraces with grassed-waterway outlets (Don Redell, Personal Communication, 1978).

The SDR values used on the example farms are given in Table E-7.

TABLE E-6.

C FACTORS USED ON EXAMPLE FARMS

State and Soil Type	Tillage Method	CROP ROTATIONS					
		S	G	SSSS-HHHH	GGGG-HHHH	S-HHHH	G-HHHH
New York (All Spring Conservation Tillage)							
Honeoye-Lima							
	Conservation	.402	.127	.154	.069	.040	.040
	No-till	.381	.073	.142	.041	.038	.029
Lima-Kendaia							
	Conservation	.422	.165	.166	.092	.047	.047
	No-till	.410	.095	.157	.050	.045	.032

Iowa (All Conservation tillage)

	G(Sp)	G(F)	G-Sb(Sp)	C-Sb(F)	GO-HHH	GGO-HH	GGG-OH
Muscatine	.117	.124	.211	.219	.028	.047	.070
Tama	.122	.129	.211	.219	.028	.048	.071
Dinsdale	.126	.134	.228	.239	.029	.049	.074
Garwin	.122	.129	.211	.219	.028	.048	.071

Texas (All Spring Conventional Tillage)

	C	Sg	W	H	CSCg
All soils	.60	.40	.20	.05	.53
	CSgSg	CWW	SgWW	C Sg	C Sg W
All Soils	.45	.30	.23	.49	.35

Sp = Spring tillage, F = Fall tillage

S : Corn Silage
G : Corn grain
H : Hay
Sb : Soybeans
O : Oats
C : Cotton
Sg : Sorghum
W : Wheat

TABLE E-7

VALUES FOR THE SEDIMENT DELIVERY RATIO ON THE EXAMPLE FARMS

State and Soil Type	No Structures	D_1*	D_2*	T_1*	T_2*	T_3*	T_4*	T_5*
New York								
Honeoye-Lima	.30	.78	1.0	.16	.20	.20	.26	.23
Lima-Kendaia	.78	1.0	1.0	.16	.20	.20	.60	.40
Iowa								
Muscatine	.29	---	---	.20	.19	.19	.07	---
Tama	.35	---	---	.20	.20	.17	.16	---
Dinsdale	.72	---	---	.20	.20	1.0	1.0	---
Garwin	1.0	---	---	.20	.20	1.0	1.0	---
Texas								
Houston-Black	.38	---	---	.20	.20	.20	.20	---
Heiden	.75	---	---	.20	.20	.81	.15	---
Trinity	1.0	---	---	1.0	1.0	1.0	.20	---

* D represents a diversion ditch system,

T represents a terrace system. The number subscripts are keyed to specific intervals evaluated for the example farms.

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APPENDIX F

EFFECTS OF SOIL AND WATER CONSERVATION PRACTICES ON RUNOFF AND POLLUTANT LOSS FROM SMALL AGRICULTURAL WATERSHEDS: A SIMULATION APPROACH

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The effect of soil and water conservation practices (SWCPs) on the transport of pollutants is complex and not always obvious. In an attempt to better understand how SWCPs affect runoff, sediment loss, and pesticide and nutrient washoff from agricultural lands, we have conducted a study of SWCPs using the Agricultural Runoff Management (ARM) Model (Donigian and Crawford 1976; Donigian *et al.* 1977). The ARM Model is a collection of the physical processes that describes the hydrologic cycle, sediment fines generation and transport, pesticide interaction in the soil and washoff, and nutrient transformations and washoff. Thus, the ARM Model can provide the necessary scope and abundance of information to evaluate the effectiveness of soil and water conservation practices in controlling runoff, sediment loss, and the movement of pesticides and nutrients from agricultural watersheds.

Soil and water conservation practices have an impact on the hydrologic and physical characteristics of a watershed, in addition to the timing and type of agronomic practices. Specific ARM Model parameters are related to certain hydrologic and physical characteristics of an agricultural watershed, while other parameters describe agronomic practices. As these characteristics are changed for a specific SWCP, the model parameter values can be adjusted to reflect the use of the SWCP. Output from the model for different SWCPs is compared to the corresponding output for the conventional practice or base condition. The total volume (or mass) and frequency of runoff, sediment loss, and pesticide and nutrient washoff are compared for the different practices. The change in frequency or volume for a particular constituent by a SWCP is used to evaluate the effectiveness of that SWCP to control that constituent. This analysis was done for three SWCPs (no tillage, contouring, and terracing with contours) on small agricultural watersheds in Watkinsville, Georgia, and East Lansing, Michigan, using 10 years of meteorologic data as input to the model. This work was performed for Cornell University under an EPA research grant to determine the effectiveness of SWCPs for pollution control.

SUMMARY OF CONCLUSIONS

The ARM Model has been used to analyze the relative effectiveness of three SWCPs (no tillage, contours, and terraces and contours) for controlling runoff, sediment loss, and pesticide and nutrient washoff from two small agricultural watersheds. The results show that, in general, the three SWCPs reduce surface runoff, sediment loss, and associated constituents. Subsurface flow is increased, as is soluble nitrogen (NO_3). Adsorbed pollutants (atrazine, methyl parathion, and soluble NH_4 and PO_4) do not exhibit this increase although their major form of transport is in solution. Sediment and associated constituents decrease for all SWCPs studied.

Terraces with contours is the most effective SWCP of the three studied for controlling most constituents. No tillage is the most effective in reducing sediment-associated pollutants. Contours is more effective in decreasing soluble constituents than no tillage; however, it is less effective than terraces plus contours.

The two watersheds studied are in different climatic regions of the United States and therefore exhibit different responses to the introduction of the three SWCPs. In general, the difference in effectiveness of the SWCPs is not large. Where major differences do occur they are a result of differing climatic conditions. The influence of snow in winter months on the P6 watershed (Michigan) is significant. Snow cover delays and dampens the hydrologic response of the watershed and often reduces what otherwise would be peak rainfall and runoff events.

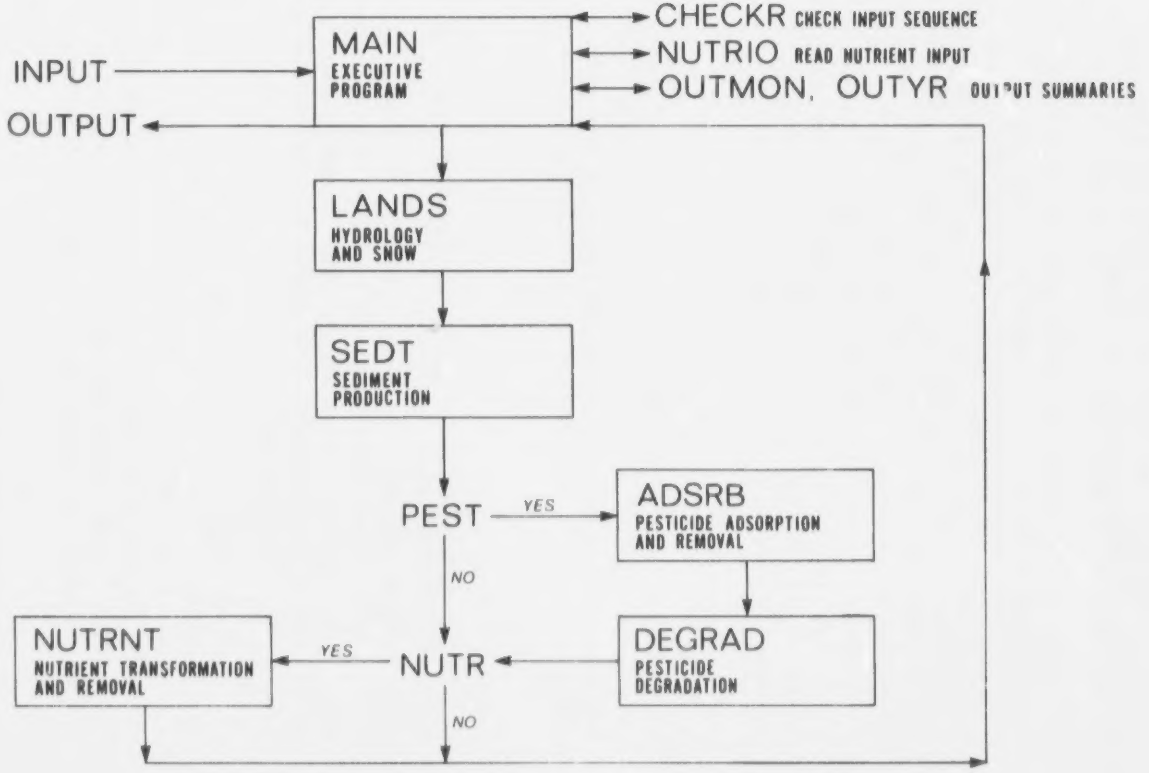
The parameter value changes in the ARM Model to represent the alternative SWCPs are estimates. As such, they and their associated algorithms are subjected to potential refinement once we have a better understanding of the actual mechanisms at work. However, the information gained from this study is useful in providing a better understanding of how the different hydrologic, sediment, pesticide, and nutrient processes interrelate and react to changes imposed by man.

AGRICULTURAL RUNOFF MANAGEMENT (ARM) MODEL

Model Structure

The ARM Model is a continuous model that simulates runoff (including snow accumulation and melt), sediment, pesticides, and nutrient contributions to stream channels from both surface and subsurface sources. Figure F-1 demonstrates the general structure and operation of the ARM Model. The major components of the model individually simulate the hydrologic response (LANDS) of the watershed, sediment production (SEDT), pesticide adsorption/desorption (ADSRB), pesticide degradation (DEGRAD), and nutrient transformations (NUTRNT). The executive routine, MAIN, controls the overall execution of the program: calling subroutines at proper intervals, transferring information between routines, and performing the necessary input and output functions.

In order to simulate vertical movement and transformations of pesticides and nutrients in the soil profile, specific soil zones (and depths) are established so that the total soil mass in each zone can be computed. Total soil mass is a necessary ingredient in the pesticide adsorption/desorption reactions and nutrient transformations. The vertical soil



MAIN
EXECUTIVE PROGRAM
INPUT
OUTPUT
CHECKR CHECK INPUT SEQUENCE
NUTRIO READ NUTRIENT INPUT
OUTMON, OUTYR OUTPUT SUMMARIES
LANDS
HYDROLOGY AND SNOW
SEDT
SEDIMENT PRODUCTION
PEST
YES
ADSRB
PESTICIDE ADSORPTION AND REMOVAL
NO
DEGRAD
PESTICIDE DEGRADATION
NUTR
YES
NUTRNT
NUTRIENT TRANSFORMATION AND REMOVAL
NO

FIGURE F-1. ARM MODEL STRUCTURE AND OPERATION zones simulated in the ARM Model include the surface, upper, lower, and groundwater zones. The depths of the surface and upper soil zones are specified by the model input parameters and are generally 2-8 mm and 75-150 mm, respectively. The upper zone depth corresponds to the depth of the incorporation of soil-incorporated chemicals. It also indicates the depth used to calculate the mass of soil in the upper zone whether agricultural chemicals are soil-incorporated or surface-applied.

The transport and vertical movement of pesticides and nutrients, as conceived in the ARM Model, is indicated in Figure F-2. Pollutant contributions to the stream can occur from the surface zone, the upper zone, and the groundwater zone. Surface runoff is the major transport mechanism carrying dissolved chemicals, pesticide particles, sediment, and adsorbed chemicals. The interflow component of runoff can transport dissolved pesticides or nutrients occurring in the upper zone. Vertical chemical movement between the soil zones is the result of infiltrating and percolating water. From the surface, upper, and lower zones, uptake and transformation of nutrients and degradation of pesticides is allowed. The groundwater zone is presently considered a sink for deep percolating chemicals.

Model Components

The algorithms or equations used to describe the processes simulated by the ARM Model are fully discussed in the original model reports (Crawford and Donigian, 1973; Donigian and Crawford, 1976; Donigian et al. 1977). A brief presentation of the general methodology is included here.

Hydrology--

Hydrologic simulation by the LANDS subprogram is derived from modifications of the Stanford Watershed Model (Crawford and Linsley, 1966) and the Hydrocomp Simulation Program (Hydrocomp, Inc., 1976). Through a set of mathematical functions, LANDS simulates continuously the major components of the hydrologic cycle, including interception, surface runoff, interflow, infiltration, and percolation to groundwater. In addition, energy balance calculations are performed to simulate the processes of snow accumulation and melt.

Sediment--

The algorithms for simulating soil loss, or erosion, were initially derived from research by Negev (1967) at Stanford University and have been subsequently influenced by the work of Meyer and Wischmeier (1969) and Onstad and Foster (1975).

Although Negev simulated the entire spectrum of the erosion process, only sheet and rill erosion are included in the ARM Model. The two component processes of sheet and rill erosion pertain to (1) detachment of soil fines (generally the silt and clay fraction) by raindrop impact, and (2) pick-up and transport of soil fines by overland flow. These processes are represented as follows:

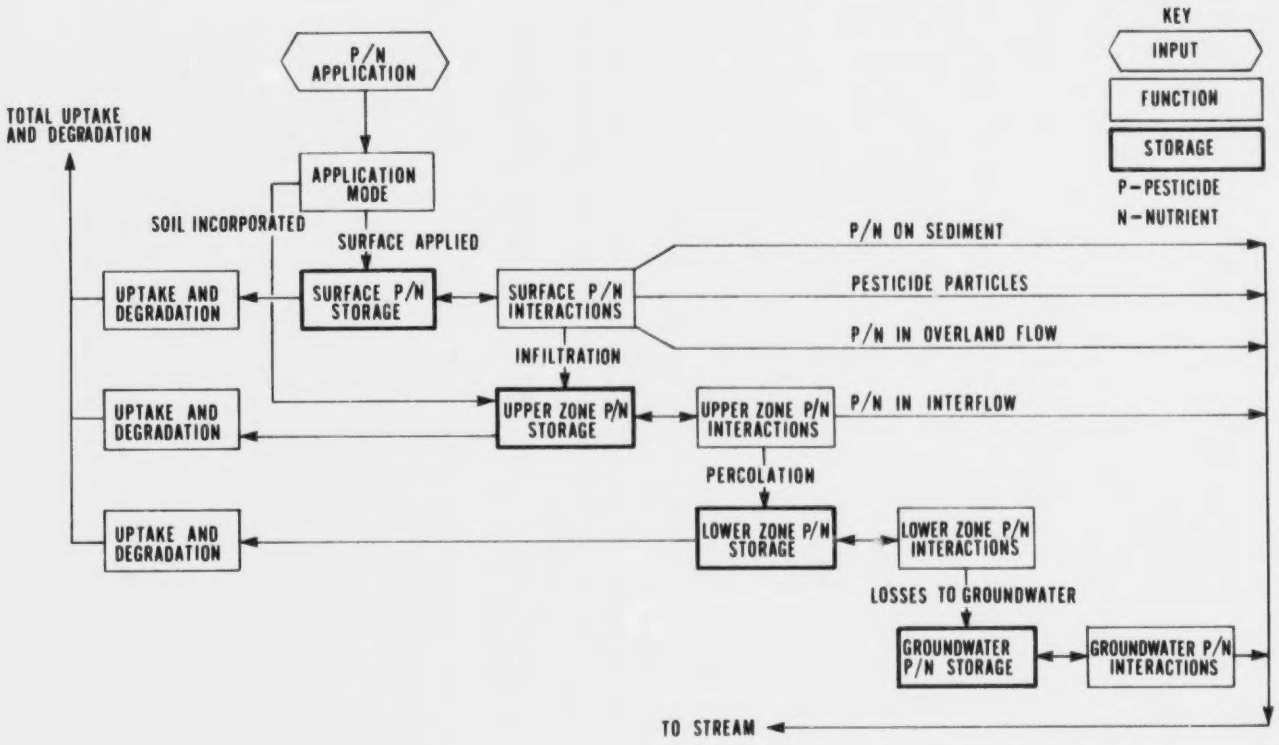

KEY
INPUT
FUNCTION
STORAGE
P—PESTICIDE
N—NUTRIENT
P/N APPLICATION
APPLICATION MODE
SOIL INCORPORATED
SURFACE APPLIED
SURFACE P/N STORAGE
SURFACE P/N INTERACTIONS
P/N ON SEDIMENT
PESTICIDE PARTICLES
P/N IN OVERLAND FLOW
INFILTRATION
UPPER ZONE P/N STORAGE
UPPER ZONE P/N INTERACTIONS
P/N IN INTERFLOW
PERCOLATION
LOWER ZONE P/N STORAGE
LOWER ZONE P/N INTERACTIONS
LOSSES TO GROUNDWATER
GROUNDWATER P/N STORAGE
GROUNDWATER P/N INTERACTIONS
TO STREAM
TOTAL UPTAKE AND DEGRADATION
UPTAKE AND DEGRADATION
UPTAKE AND DEGRADATION
UPTAKE AND DEGRADATION

FIGURE F-2. PESTICIDE AND NUTRIENT MOVEMENT IN THE ARM MODEL

Soil fines detachment:

$$RER(t) = (1-COVER(T))*SMPF*KRER*PR(t)^{JRER} \qquad (F.1)$$

Soil fines transport:

$$SER(t) = \begin{cases} KSER*OVQ(t)^{JSER}, \text{ for } SER(t) \leq SRER(t) & (F.2) \\ SRER(t), \text{ for } SER(t) > SRER(t) & (F.3) \end{cases}$$

$$ERSN(t) = SER(t)*F \qquad (F.4)$$

where RER(t) = soil fines detached during time interval t, metric tons/ha
COVER(T) = fraction of vegetal cover as a function of time, T within the growing season
KRER = detachment coefficient for soil properties
SMPF = supporting management practice factor (P-factor in Universal Soil Loss Equation, Wischmeier and Smith 1965)
PR(t) = precipitation during the time interval, mm
JRER = exponent for soil detachment
SER(t) = transport of fines by overland flow, metric tons/ha
JSER = exponent for fines transport by overland flow
KSER = coefficient of transport
SRER = reservoir of soil fines at the beginning of time interval, t, metric tons/ha
OVQ(t) = overland flow occurring during the time interval, t,mm
F = fraction of overland flow reaching the stream during the time interval, t
ERSN(t) = sediment loss to the stream during the time interval, t, metric tons/ha

In the operation of the algorithms, the soil fines detachment (RER) during each time (5 or 15 min) interval is calculated by Equation F.1 and added to the total fines storage or reservoir (SRER). Next, the total transport capacity of the overland flow (SER) is determined by Equation F.2. Sediment is assumed to be transported at capacity if sufficient fines are available, otherwise the amount of fines in transport is limited by the fines storage, SRER (Equation F.3). The sediment loss to the waterway in the time interval is calculated in Equation F.4 by the fraction of total overland flow that reaches the stream. An overland flow routing technique determines the flow contribution to the stream in each interval. After the fines storage (SRER) is reduced by the actual sediment loss to the stream (ERSN), the algorithms are ready for simulation of the next time interval. Thus, the sediment that does not reach the stream is returned to the fines storage and is available for transport in the next time interval. The methodology attempts to represent the major processes of importance in soil erosion so that the impact of land management practices (for example, tillage, terracing, mulching, etc.) can be specified by their effects on the sediment parameters.

Since land cover by growing crops and crop residues has a major impact on sediment loss, the variability in the land surface cover is explicitly represented in the ARM Model. The land cover variable in Equation F.1, COVER(T), represents the fraction of the land surface effectively protected from the kinetic energy and detachment capability of rainfall. Monthly cover values as of the first day of the month are specified by the user. The model interpolates linearly between the monthly values to evaluate land cover on each day. Figure F-3 demonstrates the land cover function in the model. The kinetic energy of rainfall is effectively dissipated by the land cover with values of 90 to 95 percent of the area. Thus, judicious use of land cover function allows simulation of various land surface conditions for different practices.

The timing and severity of tillage operations have a controlling effect on the sediment loss from an agricultural watershed. With regard to sediment production, the effect of tillage operations is to increase the mass of soil fines available for transport and produce a reasonably uniform distribution of fines across the watershed. Consequently, the ARM Model allows the user to specify the dates of tillage, planting, or other land-surface disturbing operations. For each of these dates the user must specify a new detached soil fines storage resulting from the operation. At the beginning of each tillage day the ARM Model resets the fines storage to the new value, resulting in a uniform fines distribution across the watershed. The amount of fines storage produced by different tillage operations is related to the depth and extent of the operation and edaphic characteristics.

Pesticide--

The process of pesticide adsorption/desorption onto sediment particles is a major determinant of the amount of pesticide loss that will occur. This process establishes the division of available pesticide between the water and sediment phases, and thus specifies the amounts of pesticide transported in solution and on sediment. To simulate this process in the ARM Model, the following equation is used:

$$X/M = KC^{1/N} + F/M \qquad (F.5)$$

where X/M = pesticide adsorbed per unit soil, μg/g

F/M = pesticide adsorbed in permanent fixed state per unit soil. F/M is less than or equal to FP/M, where FP/M is the permanent fixed capacity of the soil in μg/g for each pesticide.

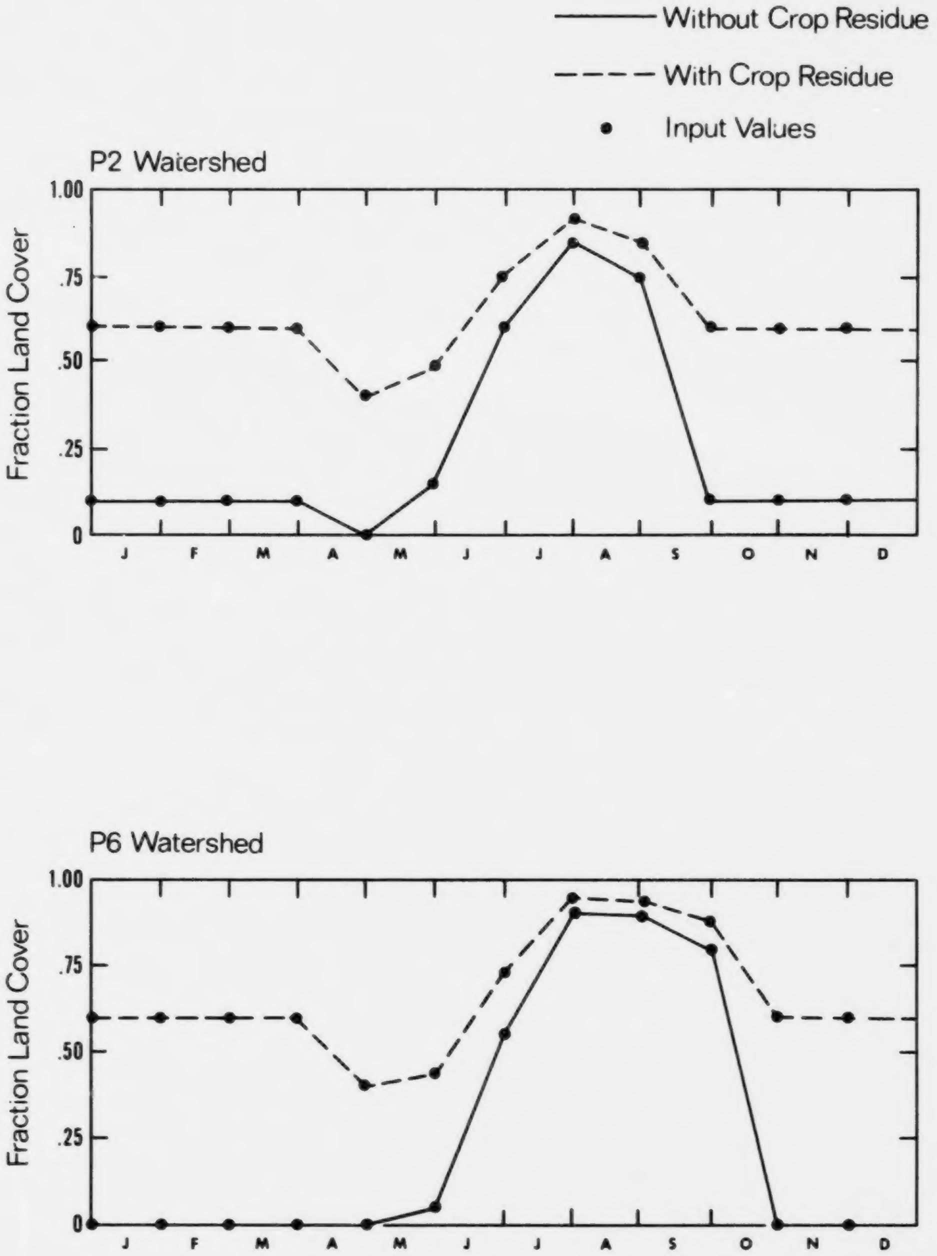
C = equilibrium pesticide concentration in solution, mg/l

N = exponent

K = coefficient

This algorithm is comprised of an empirical term, F/M, plus the standard Freundlich single-valued (SV) adsorption/desorption isotherm. The empirical term, F/M, accounts for pesticides (for example, paraquat) that are permanently adsorbed to soil particles and will not desorb under repeated washing.

The ARM Model includes an option to use a non-single-valued (NSV) adsorption/desorption function because research has indicated that the assump-



P2 Watershed

P6 Watershed

FIGURE F-3. LAND COVER WITH AND WITHOUT CROP RESIDUE tion of single-valued adsorption/desorption is not valid for many pesticides (Davidson et al. 1973). In these cases, the adsorption and desorption processes result in different pesticide concentrations. The form of the desorption equation is identical to Equation F.5 except that K and N values are replaced by K' and N' respectively, with the prime denoting the desorption process. The user specifies the N' value as an input parameter and the ARM Model calculates K' as a function of the adsorption/desorption parameters (K, N, N') and the pesticide solution concentration (Davidson et al. 1973). The NSV function simulates higher pesticide concentrations on sediment than the SV function in order to represent the irreversibility of the adsorption process.

Attenuation of applied pesticides, through volatilization and degradation processes, is critical to the accurate simulation of pesticide runoff because these mechanisms control the amount of chemical available for transport. These processes are not well understood and are topics of continuing research. To approximate the pesticide attenuation following application, the ARM Model includes a step-wise first-order attenuation function that allows the use of different degradation rates for separate time periods after application. The function calculates the combined degradation of pesticides by volatilization, microbial degradation, and other attenuation mechanisms. This approach was chosen after evaluating both simpler and more sophisticated degradation models (Donigian et al. 1977).

The simulation of methyl parathion washoff requires the addition of algorithms to represent pesticide storage, decay, and washoff from the crop cover. All previous pesticides studied with the ARM Model are applied to the soil surface at planting time. Methyl parathion, an insecticide used on cotton, is applied by spraying the crop every one to three weeks during the crop's growing season once the cotton squares begin to form. The amount of methyl parathion applied on the crop compared to the amount missing the crop and reaching the soil surface is assumed to be proportional to the fraction of crop cover. For example, if the crop cover equals 0.9, then nine-tenths of the pesticide applied by spraying is stored on the crop and one-tenth is stored on the surface layer of the soil.

Pesticide is removed from the crop by rainfall according to the following relationship:

$$PRFC = CSTR*(1.0 - EXP(PCK*PR*TIMFAC/60.)) \qquad (F.6)$$

where PRFC - pesticide removed from crop storage, grams
CSTR = pesticide stored on crop, grams
PCK = pesticide crop constant
PR = rainfall, mm/interval (min)
TIMFAC = interval time, min

This equation assumes that the amount of pesticide washed off the crop is exponentially related to the rainfall intensity. Data to confirm this equation and determine PCK are scarce. Information on parathion washoff collected by Gunther (1977) was the most pertinent of the literature reviewed.

Pesticide washed off the crop cover by rainfall is added to that in the soil surface layer. In the following time intervals this pesticide now in the soil surface layer can either stay in solution, adsorb onto sediment particles, or infiltrate into the upper and lower soil zones.

The remaining pesticide on the crop is decreased by decay or degradation. Degradation on the crop is a first-order decay process, same in concept as degradation in the surface, upper, and lower soil zones. However, the daily decay rate for the pesticide stored in the crop cover can be different from the rate used for pesticide in the soil. This is because of the different mechanisms involved in breaking down the pesticide on the crop and in the soil. Degradation of methyl parathion in the soil is mainly by biological organisms; methyl parathion's half-life in the soil is 45 days (Menzie, 1972). On the crop surface methyl parathion is degraded by photochemical processes and resulting half-life is only 8 days (Brown, EPA, personal communication, 1978). The ARM Model uses each degradation rate for methyl parathion, one for the pesticide on the crop and the other for the pesticide in the soil.

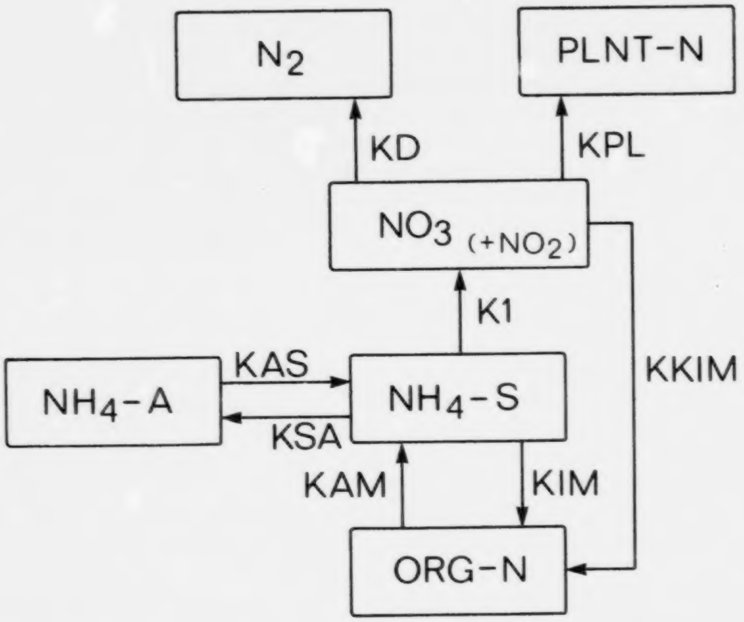
Finally, any methyl parathion remaining on the crop at harvest time is removed in proportion to the percent of crop cover removed by the harvest. This allows for complete accounting of the pesticide by the model at all stages of its existence in the watershed.

Nutrients--

Nutrient simulation in the ARM Model attempts to represent the reactions and transformations of nitrogen and phosphorus compounds in the soil profile as a basis for predicting the nutrient content of agricultural runoff. The nutrient model assumes first-order reaction rates and is derived from work by Mehran and Tanji (1974), and Hagin and Amberger (1974). The processes simulated include immobilization, mineralization, nitrification/denitrification, plant uptake, and adsorption/desorption. In the model, fertilizer, plant residue, or animal waste is applied in their chemical form (NH_4, NO_3, PO_4, Cl, and organic N and P) to the land surface or into the soil.

The ARM Model simulates nutrient movement in the watershed by water or sediment. Transformations of nutrients determine the nutrient forms in each soil zone and their resulting susceptibility to movement. The ARM Model imitates nutrient transport processes only to the extent that is needed to predict runoff quality and quantity. The model does not consider the generally secondary movement of soil chemicals by concentration and thermal gradients. However, it does model the lateral and downward transport of chemicals by water from the soil zones. Lateral transport of nutrients towards the stream can occur from the surface and upper zone storages. Groundwater transport is not currently modeled.

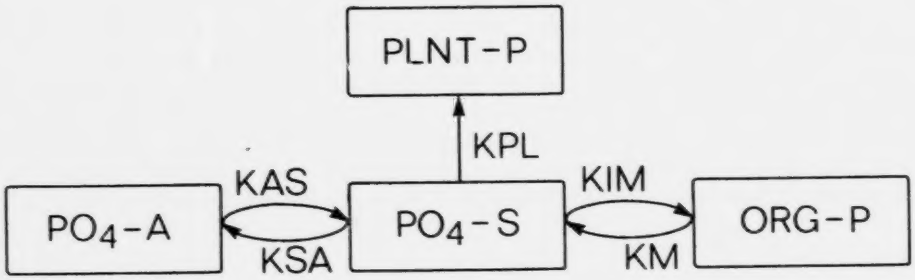
Figure F-4 diagrams the transformation pathways and storages and gives the names of the reaction rate input parameters. Reaction rates are input on a per day basis for each soil zone. Nitrite (NO_2) transforms so quickly in most agricultural areas that it is not considered separately. The adsorbed phase represents the nutrients in a complex form along with those adsorbed on the soil. The plant uptake rates (KPL) are modified monthly by


N_2
PLNT-N
KD
KPL
NO_3 $(+NO_2)$
K1
KKIM
KAS
NH_4-A
NH_4-S
KSA
KAM
KIM
ORG-N

A. Nitrogen transformations in ARM model

Key

N - Nitrogen
P - Phosphorus
A - Adsorbed
S - Solution
K - Reaction Rate Parameters


PLNT-P
KPL
KAS
KIM
PO_4-A
PO_4-S
ORG-P
KSA
KM

B. Phosphorus transformations in ARM model

FIGURE F-4. NUTRIENT TRANSFORMATIONS

an input parameter which depends on the stage of crop growth. These monthly input parameters are adjusted to represent the crop uptake of N and P from the soil storages and to distribute it throughout the growing season.

Soil temperature is presently the only environmental factor that is modeled as affecting the reaction rates with the exception that the transformations are stopped entirely at very low moisture levels. Soil temperatures for the surface and upper zone are determined by regression equations based on air temperature, while the daily soil temperature for the lower zone and groundwater is interpolated from average monthly input values. Other factors deemed as having a constant influence during the simulation period, such as soil pH, are represented by adjustments to the input reaction rates.

DESCRIPTION OF SOIL AND WATER CONSERVATION PRACTICES

In a joint study by the EPA and ARS (Stewart et al. 1975), 18 soil and water conservation practices (SWCPs) were determined to be important in controlling runoff and sediment loss from agricultural areas. Of these 18 practices we selected three for detailed study. These three practices are the use of contours, terraces and contours, and no tillage. These practices were selected based on their relative importance as a SWCP and the ability of the model to represent the particular changes involved in each practice.

To represent a SWCP by changing model parameter values, it is important to define how the SWCP differs from the base conditions or standard practice. In addition, definition of the base conditions is necessary, as the base conditions change for each region of the country. The ARM Model has been tested on the ARS-EPA P2 watershed near Watkinsville, Georgia, and the MSU-EPA P6 watershed in East Lansing, Michigan, (Donigian et al. 1977) and was found capable of representing the hydrology, sediment movement, and pesticide and nutrient washoff measured on the two watersheds. This previous work provided initial parameter values for the model. It also provided an initial practice on which to base these parameter values. A comparison of the management practices used on the watershed with three SWCPs to be studied helped to define the necessary parameter value changes. But before the parameter changes can be discussed it is important to clearly define each soil and water conservation practice relative to the base conditions.

Base Conditions

For this study, the base conditions represent conventional practices in common use in the region of the test watershed. For our two watersheds this means planting row crops in straight rows parallel to (that is, up and down) the land slope. Tillage of the soil is in the form of discing and is done in the spring to prepare the soil for planting. After harvest the crop residue is removed from the field and the soil is fallow through the winter until a new cro is planted in the spring.

No Tillage

The no tillage SWCP differs from the base conditions in two ways. There is no tillage prior to spring planting and the crop residue is left on the field after harvest. Planting is done with the crop residue remaining from the previous fall. This protects the soil from erosive forces until the coming spring when a new crop is planted.

Contours

The contour SWCP differs from the base conditions only in that the crops are planted in rows perpendicular to the slope of the land, which by definition are contoured rows. For the purposes of this study, the contour SWCP is assumed to use discing in the spring prior to planting and removal of crop residue after harvest.

Contours and Terraces

The terrace SWCP requires physical alteration of the land to establish and maintain terraces. Both the P2 and P6 watersheds are assumed to be divided into two terraces if this SWCP is used. Contoured rows are used with the terraces. However, it is assumed that discing and crop residue removal practices are retained from the base condition practice.

These three SWCPs plus the base condition are modeled by a selection of parameter values which represent each watershed condition. How the parameters and their values were selected for each SWCP is discussed in the following pages.

ARM MODEL PARAMETER VALUES AND SWCPs

The selection of the ARM Model parameters and their values for different SWCPs were based on the expected physical and agronomic changes associated with a particular SWCP. A total of eight input parameters for which adjustments could be reasonably estimated were identified as being related to SWCPs. These parameters and values for each watershed condition are listed in Table F-1.

Determining how to change parameter values to represent a particular SWCP on a watershed is often difficult. For some characteristics we have related parameter value changes to established techniques (for example, Universal Soil Loss Equation, SCS Curve Number Runoff Method) derived from field data. However, for certain parameters, our own experience, judgment, and knowledge of the physical processes and their representation in the model was our only guide. The general procedure was to estimate initial parameter changes from the literature (which was non-existent for some parameters) and then adjust the values to conform with our experience on other watersheds. These estimating procedures allowed us to establish parameter values for each of the three SWCPs included in this study.

Upper Zone Nominal Moisture Storage

UZSN (upper zone nominal moisture storage) can be related to the initial abstraction parameter, Ia, as defined by the SCS procedures for estimating storm runoff volume (Chow 1964). The relationship is assumed to be propor-

TABLE F-1. ARM MODEL PARAMETER VALUE CHANGES FOR SWCPs

		Soil and Water Conservation Practices		
Parameters[a]	Base Condition	No Tillage	Contours	Terraces Contours
P2 Watershed				
UZSN	0.42	0.42	0.50	0.65
NN	0.20	0.32	0.25	0.25
L	100	100	100	250
SS	0.025	0.025	0.025	0.015
COVPMO	(see Figure F-3)			
SMPF	1.0	1.0	0.5	0.35
KSER	0.6	0.55	0.5	0.5
SRERTL[b]	1.5	0.15	1.5	1.5
P6 Watershed				
UZSN	0.17	0.17	0.20	0.26
NN	0.20	0.32	0.25	0.25
L	60	60	60	200
SS	0.06	0.06	0.06	0.018
COVPMO	(see Figure F-3)			
SMPF	1.0	1.0	0.5	0.35
KSER	0.6	0.55	0.5	0.5
SRERTL[c]	1.0	0.10	1.0	1.0

[a] Parameters are defined as follows:

UZSN = nominal upper zone moisture storage, in.
NN = Manning's n (roughness) for overland flow
L = length of overland flow path, ft.
SS = slope of overland flow path
COVPMO = fraction of land covered by vegetation, monthly value
SMPF = supporting management practice factor (equal to the P factor in the USLE)
KSER = sediment washoff coefficient
SRERTL = sediment fines produced by tillage operations, tons/acre

[b] Tillage occurs once each year on day 115 (April 25) on P2 for the 10-yr. simulation.

[c] Tillage occurs once each year on day 139 (May 19) on P6 for the 10-yr. simulation.

tional: UZSN should be changed by the same percent as Ia when comparing a SWCP with a base condition. This was done for each of three antecedent moisture conditions included with SCS method. The average percent change for a particular SWCP was used to determine the UZSN value for that SWCP. The calibrated UZSN values for the P2 and P6 watersheds with contours were the baselines from which the other values were calculated.

Manning's Roughness Coefficient

NN (Manning's roughness coefficient for overland flow) is proportional to the amount and type of ground cover and soil type. An inexact, but relative, value of NN can be found from the literature (Chow 1964) for comparison of SWCPs with the base condition value. However, the NN values in Table F-1 are based largely on our own experience and judgment.

Length of Overland Flow

L (length of the overland flow path) changes only for the terrace SWCP when compared to the base condition. The modification of the watershed with terraces changes the overland flow path such that instead of draining to a central location to form a stream channel the flow must instead cross the width of the terrace and then traverse the length of the terrace to its outflow point. This increase in length will vary with the terrace configuration. For our use on the P2 watershed the addition of two terraces increased L from 30.5 m (100 ft) to 76.2 (250 ft). On the P6 watershed, L changed from 18.3 m (60 ft) to 61.0 m (200 ft) with the addition of two terraces.

Slope of Overland Flow Path

SS (slope of the overland flow path) is affected only by the terrace SWCP. Installation of terraces both increases the length of the flow path and decreases the slope. For this specific situation the slope of the P2 watershed is decreased from 0.025 to 0.015. For the P6 watershed, the slope is changed from 0.06 to 0.018.

Crop Cover

COVPMO (cover per month) is the fraction of the land surface covered by both live and dead vegetation and effectively protects the soil surface from the erosive force of raindrops. When crop residue is left on the watershed after harvesting, as is done with the no tillage SWCP, the fraction of land covered each month is equal to values measured by a joint EPA-ARS study (Smith *et al*. 1977). Monthly cover values for the period between harvest and planting are close to zero for conditions where the crop residue is removed and the land left without cover. On the P2 watershed, a minimum cover value of 0.1 was used as a reasonable value for this situation (Figure F-3).

Supporting Management Practice Factor

SMPF (supporting management practice factor) has been added to the ARM

Model to provide a means of incorporating the effect of management practices on the generation of sediment fines (Equation F-1). SMPF is equal to the P-factor of the USLE. As such, data have been collected from which SMPF can be calculated for the contour and terrace SWCPs (Stewart et al. 1975).

Sediment Washoff Coefficient

KSER (sediment washoff coefficient) is affected by changes to soil surface conditions brought about by the introduction of SWCPs. Fleming and Leytham (1976) show a relationship between KSER, Manning's roughness coefficient, overland flow length and slope, and mean surface particle size, although the relationship's reliability is unknown. The first three of these factors are related to the SWCPs we are investigating. Thus, although we cannot establish any direct correspondence between KSER and a particular SWCP, relative changes can be estimated from past experience with the model. KSER is expected to decrease with practices that retard overland flow and include crop resiudes. Adjustments to KSER similar to the changes in Table F-1 were included in a previous study (Hydrocomp 1978) and were found to produce reasonable results.

Soil Fines Produced by Tillage

SRERTL (soil fines produced by tillage) is affected by the no tillage SWCP in much the same way as COVPMO. Model testing on watershed with similar practices provided SRERTL values for spring discing of the P2 and P6 watersheds. Discing is a conventional practice for a number of agricultural regions of the United States. Thus, for our study it represents the base condition. With no tillage, sediment fines are only produced by planting in the crop residue. SRERTL for the no tillage SWCP is reduced to one-tenth of the base condition value. A comparison of sediment loss data from similar watersheds of which one was tilled and one was not supports this reduction of SRERTL (Smith et al. 1977).

Nutrient Applications

Nitrogen and phosphorus transport is directly affected by the runoff and sediment parameter values adjusted for SWCPs. Removal of adsorbed phosphate, ammonium, organic nitrogen, and phosphate phosphorus is influenced directly by the amount of soil eroded, while removal of ammonium and phosphate in solution is controlled by the runoff.

In addition, SWCPs will have subtle changes on the soil nutrient transformation processes and storages. Residue incorporation will affect the mineralization and immobilization rates as well as the soil organic content. Changes in soil organic matter content will influence the adsorptive capacity. Varying practices will affect crop growth resulting in different rates of plant nutrient uptake. There are obviously many more impacts which affect transformations which feedback on the runoff and erosion processes. No present erosion and hydrologic model considers all of these interactions, and for our purposes they are assumed to be minor.

In our evaluation of SWCPs, only the change to the soil organic nitrogen and phosphorus storages under differing residue management is considered. The need for this change is obvious. Residue is left on the watersheds under no tillage practice, so more organic nitrogen and phosphorus is available to be removed on sediment. The other practices assumed that only a minimum of debris remains. Numerical estimates of stubble and root organic nitrogen and phosphorus additions to the surface and upper zones at time of harvest are given in Table F-2. These values were used in the ARM Model simulations on P2 and P6 for the different SWCPs.

WATERSHED SIMULATION AND METHODOLOGY

P2 Watershed, Watkinsville, Georgia

The P2 watershed is a small (1.3 hectare), non-terraced agricultural watershed near Watkinsville, Georgia (Figure F-5). From 1973 through 1975 EPA and ARS intensively monitored this watershed (Smith et al. 1977). The data collected by these agencies made possible a thorough calibration of the ARM Model's hydrology and sediment algorithms and testing of the pesticide and nutrient routines in the model. This has been done (Donigian et al. 1977) and the calibrated model was found to reasonably represent the hydrology, sediment loss, and pesticide and nutrient washoff of P2.

The management of the P2 watershed during the EPA-ARS study was a mixture of SWCPs. Corn was grown on the watershed in contoured rows. The watershed was disced each spring prior to planting, but in the autumn after harvest the residue was left on the field until the coming spring. The ARM Model was calibrated to represent this combination of the contour and some no tillage practices; the calibrated parameter values reflect these conditions. Thus, the calibrated parameter values are not identical to the base condition values (which assume no SWCP) or any particular SWCP. However, with knowledge of how the actual watershed management practices differ from the base conditions and the SWCPs, the appropriate parameter adjustments for each condition were evaluated.

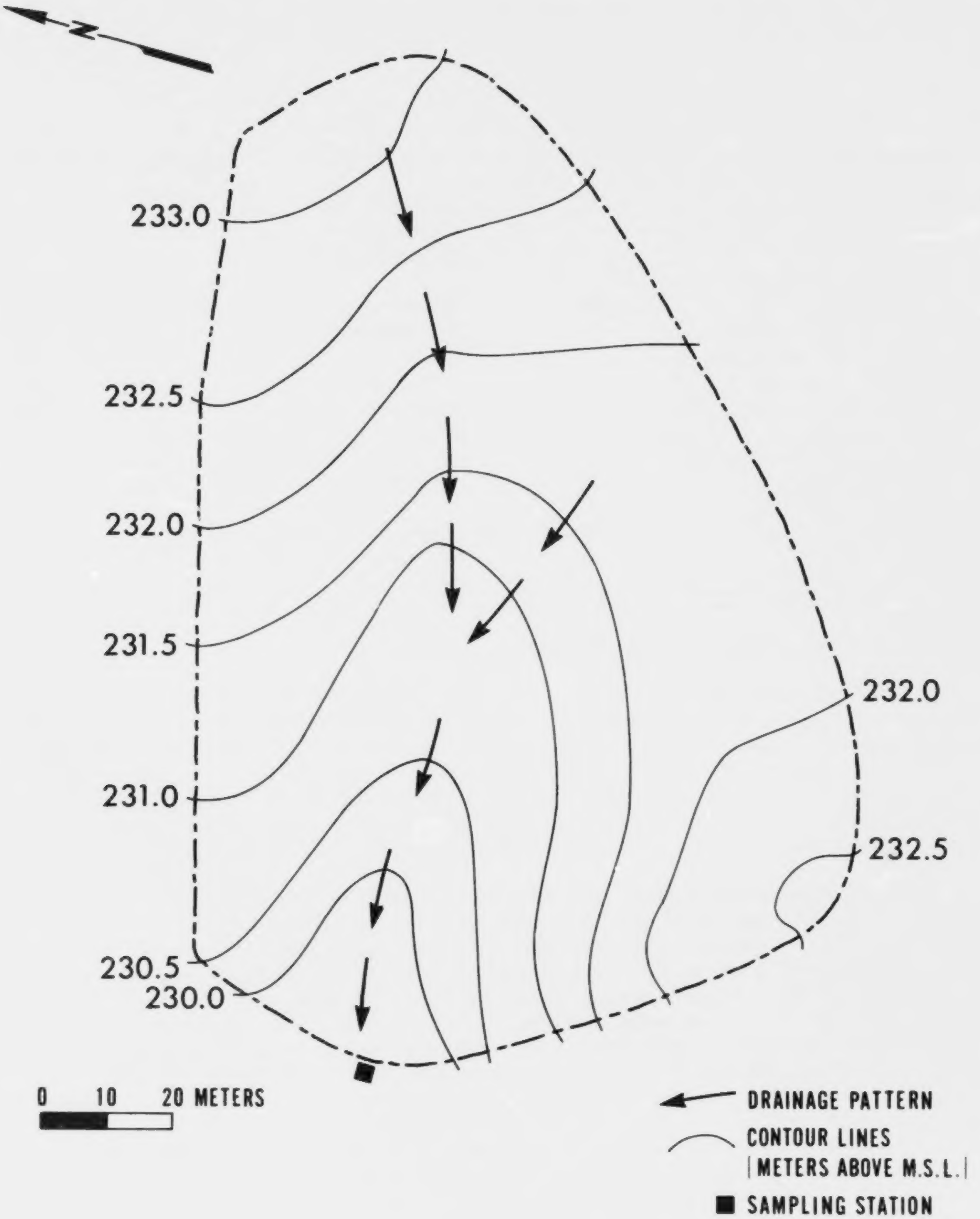
To study the effects of SWCPs on runoff, sediment loss, and pesticide and nutrient washoff the P2 watershed was simulated for the 10-yr period of 1966 through 1975. Simulation of hydrology, sediment, and pesticides by the ARM Model requires 5 or 15-min rainfall and daily evaporation. Nutrient simulation further requires daily max-min air temperature. Hourly rainfall data collected by the National Weather Service (NWS) at the nearby Athens Airport were disaggregated to 15-min amounts for use in the model. Disaggregation of hourly to 15-min rainfall uses a technique developed by Kraeger (1971) where recorded 15-min data are divided into categories by the season and size of storm. A maximum of ten different recorded storms are placed in each category. Recorded hourly rainfall to be disaggregated is matched by season and size of storm to the 15-min data. Within each category one of the ten storms is randomly selected to provide the distribution that divides the hourly data into the four 15-min values. This is done for the entire 10-yr period of hourly rainfall using the 29 months of 15-min rainfall to develop the categories and distributions. Daily evaporation and max-min air temperature data used in the simulation were also obtained from the NEW and were collected at stations within a 100-kilometer radius of the watershed.

Three different pesticides (atrazine, paraquat, and methyl parathion) were selected for study on the watershed. Atrazine and paraquat were applied and monitored during the EPA-ARS study and reasonable simulation results were obtained with the ARM Model. In the 10-yr simulation period of this study, atrazine and paraquat were applied once in the spring of each year at planting time. The application amounts (2.91 kg/ha for atrazine, 2.52 kg/ha for paraquat) and application date were selected based on average values and dates from the 3-yr EPA-ARS study. Methyl parathion was simulated with six applications of 1.12 kg/ha each during the crop growing season of each year. Fertilizer (N and P) was applied twice each year (Table F-3).

The ARM Model simulated hydrology, sediment, and pesticides on a 15-min time step. Nutrient washoff was also computed on a 15-min time step although the nutrient transformations in the soil were computed once every 6 hr. When runoff greater than a minimum value (0.003 m^3/s) occurred a complete list of runoff constituents was written to a computer file for later analysis. Separate simulation runs were made for atrazine, paraquat, methyl parathion, and nutrients for each of the three SWCPs and the base conditions.

TABLE F-2. NUTRIENT APPLICATION CHANGES FOR SWCPs

	Base Condition	No Tillage	Contours	Contours and Terraces
P2 Watershed				
Day 297				
Surface Zone				
Organic N (kg/ha)	2.52	8.96	2.52	2.52
Organic P (kg/ha)	0.34	1.68	0.34	0.34
Upper Zone				
Organic N (kg/ha)	22.4	22.4	22.4	22.4
Organic P (kg/ha)	4.48	4.48	4.48	4.48
P6 Watershed				
Day 273				
Surface Zone				
Organic N (kg/ha)	2.52	6.72	2.52	2.52
Organic P (kg/ha)	0.28	1.34	0.28	0.28
Upper Zone				
Organic N (kg/ha)	17.9	17.9	17.9	17.9
Organic P (kg/ha)	3.36	3.36	3.36	3.36


N
233.0
232.5
232.0
231.5
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230.5
230.0
232.0
232.5
0 10 20 METERS
DRAINAGE PATTERN
CONTOUR LINES
(METERS ABOVE M.S.L.)
■ SAMPLING STATION

FIGURE F-5. P2 WATERSHED, WATKINSVILLE, GEORGIA (1.3 ha)

TABLE F-3. NITROGEN AND PHOSPHORUS FERTILIZER APPLICATIONS

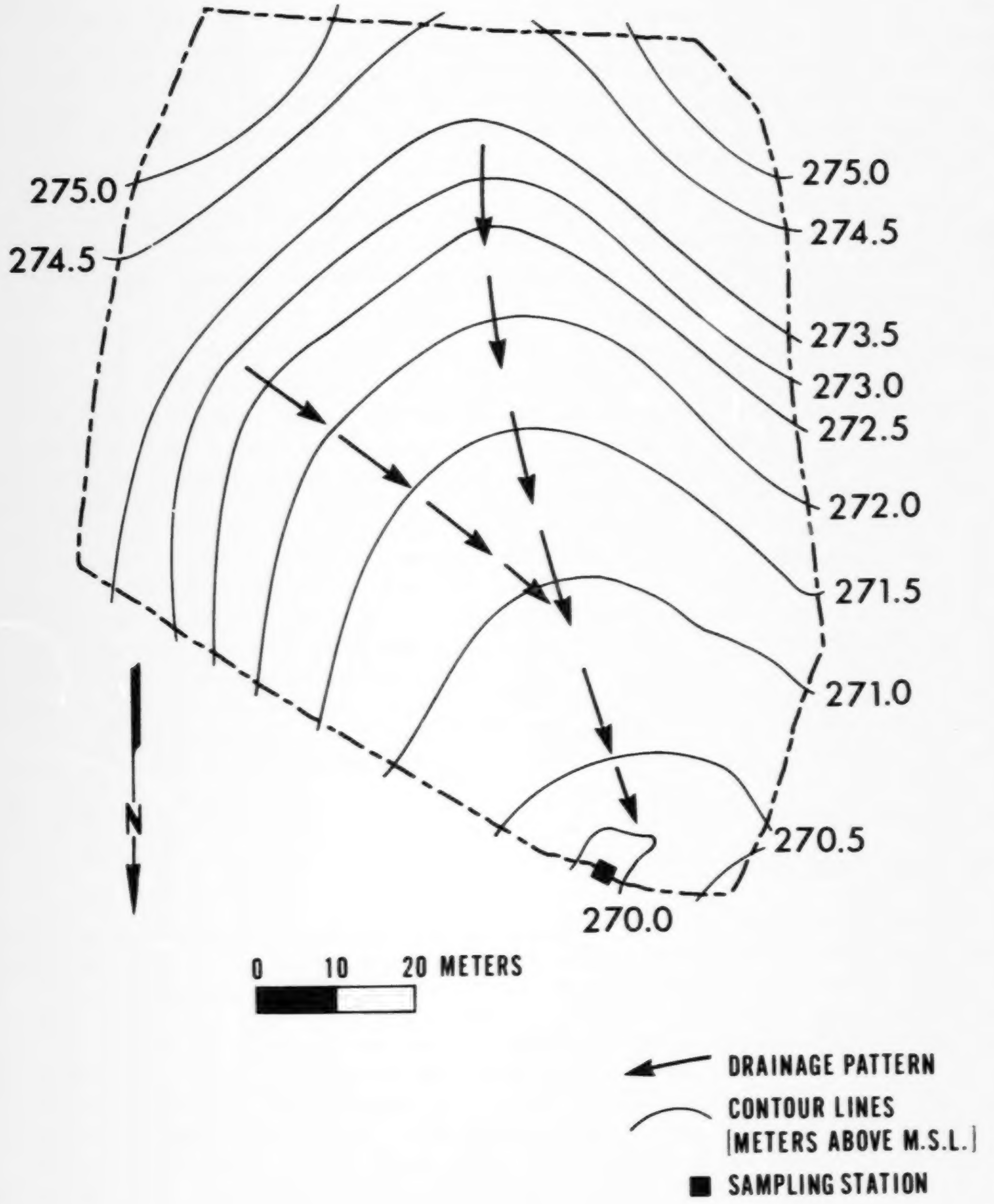
Watershed	Day of Application Each Year	Form Applied	N (kg/ha)	P (kg/ha)
P2	119	sulphate of ammonia superphosphate	38.0	33.0
	162	50% urea 50% ammonia	100.7	0.0
P6	140	ammonium nitrate monocalcium phosphate	68.3	92.9
	189	ammonium nitrate	129.9	0.0

P6 Watershed, East Lansing, Michigan

The P_6 watershed (0.8 hectares) is a non-terraced agricultural watershed located on the soil science farm at Michigan State University in East Lansing, Michigan (Figure F-6). Between 1973 and 1975, MSU monitored this watershed for an EPA-sponsored study (Ellis et al. 1978). The data collected during this study were later used to calibrate the hydrology and sediment algorithms of the ARM Model and test the pesticide and nutrient routines in the same manner as was done on the P2 watershed (Donigian et al. 1977).

Like the P2 watershed, the management of P6 consisted of a mixture of SWCPs. During the 1974 and 1975 crop seasons (the period on which the ARM Model was calibrated), corn was grown in contoured rows. The soil was tilled each spring to a depth of 7.6 cm before planting and fertilizer application (Hubbard 1975). After harvest in the autumn the watershed was left bare of residual until the new crop emerged in the spring. The ARM Model was calibrated to represent these practices, and thus the parameter values differ from the base condition values. Values were then changed to represent base conditions in the same manner as was done on the P2 watershed.

The P6 watershed was simulated for the 10-yr period of 1966 through 1975. Simulation of hydrology on this watershed included the simulation of snowfall and snow accumulation and melt during the winter months. Snow simulation requires daily wind, radiation, and dewpoint data in addition to 15-min precipitation, daily evaporation, and daily max-min air temperature required for all months. Hourly precipitation and daily meteorological data at Lansing and East Lansing were obtained form the NWS. Hourly


275.0
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N
0 10 20 METERS
DRAINAGE PATTERN
CONTOUR LINES
[METERS ABOVE M.S.L.]
■ SAMPLING STATION

P6 WATERSHED, EAST LANSING, MICHIGAN (0.8 ha) precipitation data were disaggregated into 15-min amounts in the same manner as was done for the P2 watershed.

Atrazine and paraquat were simulated on the P6 watershed. These pesticides were applied and monitored during the MSU study and were adequately simulated with the ARM Model. Each pesticide was applied in the spring of each year at planting time. The application amounts for atrazine (2.8 kg/ha) and paraquat (1.46 kg/ha) and the application date were selected based on average values and dates from the MSU study. In a like fashion fertilizer was applied on the watershed (Table F-3).

The simulation interval, criteria for writing runoff constituents to a computer file, and the number of simulation computer runs for P6 was the same as was done for P2, except that methyl parathion was not simulated on the P6 watershed.

SIMULATION RESULTS AND DISCUSSION

The continuous information produced by the 10-yr ARM Model simulations was analyzed in two ways: (1) mean annual runoff and pollutant losses, and (2) frequency of occurrence of runoff rates, pollutant concentrations, and pollutant flux (that is, mass removal in mass per minute). The SWCPs are represented only by the parameter changes discussed above. Other effects of SWCPs, such as changes in infiltration characteristics, crop growth, soil fertility, etc. are not considered due to lack of sufficient evidence to indicate appropriate parameter adjustments. Also, the pesticide and nutrient parameters (for example, application rates and time, adsorption coefficients, reaction rates) were the same for all conditions to provide a basis for comparing the effects of specific SWCPs. Thus, increasing pesticide applications to compensate for no tillage practices was not included in this analysis, but such a practice can be evaluated in a similar manner.

Mean Annual Values

P2 Watershed (Georgia) Results--

Table F-4 summarizes the mean annual runoff and pollutant losses (in solution and on sediment) for each watershed condition on the P2 watershed and provides the percent change for each SWCP from the base conditions. The results show that the total annual runoff decreases with each SWCP, with minor reductions (<10%) for no tillage, 10 to 12 percent for contours, and 30 to 40 percent reductions for contours and terraces. Since overland flow comprises more than 90 percent of the total runoff for this small watershed, the reductions for overland flow are similar. Interflow, which travels subsurface for part of its flow path, increases 10 to 12 percent for no tillage, less than 5 percent for contours, and approximately 30 percent for contours and terraces. Thus, the practices that retard and reduce overland flow also increase the subsurface flow. Both interflow and groundwater contributions increase although groundwater is not present in the runoff from this small watershed.

Sediment loss for each SWCP partially reflects the reductions in surface

TABLE F-4. P2 MEAN ANNUAL RUNOFF AND POLLUTANT LOSSES

	Mean Annual Value				Percent Change from Base		
	Base Conditions	No Tillage	Contours	Terraces and Contours	No Tillage	Contours	Terraces and Contours
Total Runoff (mm)	198.6	185.8	177.4	127.7	- 6	-11	-36
Overland Flow (mm)	185.6	171.3	163.9	110.8	- 8	-12	-40
Interflow (mm)	13.0	14.5	13.5	16.9	+12	+ 4	+30
Sediment Loss (metric tons/ha)	6.33	3.30	3.99	2.84	-48	-37	-55
Total Atrazine Loss (g/ha)	53.5	47.2	40.9	19.0	-12	-24	-64
in solution (g/ha)	53.1	46.9	40.6	18.8	-12	-24	-65
on sediment (g/ha)	0.41	0.34	0.28	0.16	-17	-32	-61
Total Methyl Parathion Loss (g/ha)	310.7	263.7	256.5	154.7	-15	-17	-50
in solution (g/ha)	283.1	245.6	241.2	145.2	-13	-14	-49
on sediment (g/ha)	27.6	18.1	15.3	9.5	-34	-45	-66
Paraquat on sediment (g/ha)	340.6	187.5	225.0	162.5	-45	-34	-52
Nitrogen (kg/ha)							
Organic	7.72	7.01	5.89	4.60	- 9	-24	-40
NH_4 - solution	2.08	2.29	1.98	1.95	+10	- 5	- 6
NH_4 - adsorbed	0.75	0.44	0.51	0.37	-41	-32	-51
NO_3 + NO_2	2.90	3.34	3.11	3.91	+15	+ 7	+35
Phosphorus (kg/ha)							
Organic	0.88	0.87	0.66	0.50	- 1	-25	-43
PO_4 - solution	0.48	0.56	0.47	0.46	+17	- 2	- 4
PO_4 - adsorbed	2.46	1.64	1.85	1.42	-33	-25	-42

runoff. Contours and terraces reduce sediment loss from 6.3 metric tons/hectare for base conditions to 2.8 metric tons/hectare, a 55 percent reduction. The sediment reduction for no tillage is 48 percent; for contours it is 37 percent. No tillage reduces sediment loss by minimizing the disruption of the soil surface and protecting the surface with crop residues remaining on the watershed. In contrast, contours reduce sediment loss by reducing overland flow.

Pesticide and nutrient contents of runoff for each SWCP are the result of the changes in runoff and sediment loss discussed above. Although atrazine is detected both in solution and on sediment, the solution for the SWCPs are in the same order as for total runoff; the negative values increase with no tillage, contours, and contours plus terraces. Atrazine loss on sediment is relatively minor and, as expected, the reductions are similar to those for sediment loss. Total atrazine loss is reduced by 64 percent for contours and terraces, 24 percent for contours alone, and 12 percent for no tillage.

Paraquat is a highly ionic compound that is strongly and irreversibly adsorbed to sediment particles. Thus, the values for paraquat loss are shown only for sediment in Table F-4 because paraquat is not transported in solution. For each SWCP, the percent reductions for paraquat and sediment are basically the same.

Methyl parathion behaves somewhat like both atrazine and paraquat. It easily and quickly adsorbs on sediment, yet the large fraction of its washoff is present in solution form. The percent reductions for the three SWCPs are close to those for total runoff. Total methyl parathion loss is reduced by 50 percent for contours and terraces and 15 to 20 percent for no tillage and contours.

The nutrient values in Table F-4 are presented by the separate nitrogen and phosphorus forms simulated with the ARM Model. The sediment-associated nutrients (organic N and P, adsorbed NH_4, and adsorbed PO_4) show the same relative percent reductions for contours and contours plus terraces as sediment loss. The reductions are in the range of 25 to 35 percent for contours and 40 to 50 percent for contours and terraces. The variations reflect the specific reaction and adsorption behavior of the individual components. The no tillage SWCP reduces adsorbed NH_4 and PO_4 by 30 to 40 percent. Organic N and P are reduced by less than 10 percent, however. This is a result of adding organic N and P to the surface zone in the form of crop residue, and this offsets the larger expected decrease due to the SWCP.

Both soluble NH_4 and soluble PO_4 show relatively minor changes for any of the SWCPs. Both constituents increase slightly under no tillage and decrease slightly for contours with and without terraces. Except to show the expected direction of a change, percentage changes less than 10.0 are not likely to be significant within the accuracy of the model. Thus, these SWCPs do not appear to have a major impact on the mean annual losses of these constituents.

On the other hand, NO_3 (and NO_2) loss is increased by 35 percent when terraces are used, with only minor increases under no tillage and contours. This change is a result of the increase in interflow when terraces are installed since interflow is the major transporting mechanism for NO_3. The other solution nutrients are not as drastically affected by interflow because they also adsorb onto sediment particles, and thus do not move as readily through the subsurface environment.

P6 Watershed (Michigan) Results--

The mean annual results simulated on the P6 watershed are presented in Table F-5.

The P6 results show that annual total runoff decreases 4 percent for no tillage, 6 percent for contours, and 25 percent for contours and terraces. Interflow is increased for both no tillage (4 percent) and terraces (24 percent). Unlike the other SWCPs, the introduction of the contour SWCP does not change the volume of interflow from that of the base conditions. The percent changes from base conditions for the three SWCPs for runoff are of similar magnitude as the P2 (Georgia) results. The major difference between the runoff percent changes on the P2 and P6 watersheds is the slightly less variability of the P6 percent changes in comparison to the P2 results. This is probably because the P6 watershed (Michigan) receives most of its winter precipitation as snowfall and P2 (Georgia) does not. Runoff from snowmelt is generally more gradual than that produced by rainfall. Thus, the SWCPs appear to be less effective in controlling slow snowmelt runoff than the more transitory runoff produced by rainfall.

Sediment loss is reduced by 71 percent with the use of no tillage. The reductions for contours and terraces are 21 and 32 percent, respectively. Snowfall, unlike rainfall, does not produce soil fines available for transport from the watershed. No tillage further reduces soil fines generation by increasing soil surface cover and, therefore, is much more effective for control of sediment and associated pollutants on the P6 watershed. The two effects combine to severely limit fines generation and washoff. Contours and terraces only limit soil washoff through decrease of runoff volume and velocity.

Pesticide washoff is largely a function of runoff and sediment loss, as is noted in the discussion of the P2 results. Atrazine is mainly removed in solution. Total atrazine loss is decreased 3 percent for no tillage, 20 percent for contours, and 62 percent for contours and terraces. These results are comparable to those for the P2 watershed.

The paraquat results are directly related to sediment loss, since paraquat is irreversibly adsorbed to sediment particles. The paraquat washoff results for the three SWCPs are nearly identical to the sediment loss results as shown in Table F-5.

Methyl parathion was not simulated on the P6 watershed. Methyl parathion is most commonly used on cotton to control insect damage. Cotton is

TABLE F-5. P6 MEAN ANNUAL RUNOFF AND POLLUTANT LOSSES

	Mean Annual Value				Percent Change from Base		
	Base Conditions	No Tillage	Contours	Terraces and Contours	No Tillage	Contours	Terraces and Contours
Total Runoff (mm)	190.9	182.6	179.2	144.0	- 4	- 6	-25
Overland Flow (mm)	165.1	155.8	153.5	111.9	- 6	- 7	-32
Interflow (mm)	25.8	26.8	26.7	32.1	+ 4	+ 0	+24
Sediment Loss (metric tons/ha)	3.51	1.01	2.78	2.40	-71	-21	-32
Total Atrazine Loss (g/ha)	79.2	76.8	63.0	29.8	- 3	-20	-62
in solution (g/ha)	77.4	75.7	61.7	28.9	- 2	-20	-63
on sediment (g/ha)	1.84	1.11	1.27	0.86	-40	-31	-53
Paraquat on sediment (g/ha)	114.8	36.3	93.4	81.2	-68	-19	-29
Nitrogen (kg/ha)							
Organic	4.22	2.19	3.60	3.16	-48	-15	-25
NH_4 - solution	2.14	2.52	1.93	1.87	+18	-10	-13
NH_4 - adsorbed	0.10	0.03	0.08	0.06	-70	-20	-40
NO_3 + NO_2	14.71	15.61	13.80	14.48	+ 6	- 6	- 2
Phosphorus (kg/ha)							
Organic	1.58	0.74	1.35	1.18	-53	-15	-25
PO_4 - solution	6.55	6.74	6.01	5.96	+ 3	- 8	- 9
PO_4 - adsorbed	1.56	0.58	1.31	1.17	-63	-16	-25

not a major commercial crop in upper Midwest, and therefore was not considered to be of enough interest to be simulated with applications of methyl parathion.

The nutrient results on the P6 watershed in general show the same type of results as were found for P2. The sediment associated nutrients (organic N and P, adsorbed NH_4, and adsorbed PO_4) decrease by 50 to 70 percent for no tillage, 15 to 20 percent for contours, and 25 to 40 percent for contours and terraces. These ranges are very close to the reductions measured for sediment loss for the three SWCPs.

In general, the soluble nutrients (soluble NH_4, soluble PO_4, and nitrate and nitrite) are only slightly increased for no tillage and are decreased by less than 10 percent for contours and contours and terraces. The exceptions are the increase of soluble NH_4 by 18 percent for no tillage and the decrease of the same constituent by 13 percent for contours plus terraces. The magnitude of these changes are due to the effect of crop residue for no tillage and the decrease in total runoff for contours plus terraces.

There is one noticeable difference in the P2 and P6 nutrient results. In the P6 watershed the PO_4 in solution is larger than adsorbed PO_4 simulated in the runoff. The reverse was simulated on P2. The removal of PO_4 in solution is a function of interflow, the kinetic reaction rate between soluble and adsorbed PO_4, and other factors. On the P6 watershed the proportion of runoff that is interflow is twice that from P2. Also the simulation of soluble PO_4 on the P6 watershed by Donigian, et al. (1977) resulted in more PO_4 washoff in solution than on sediment. This does not necessarily mean that the simulation of PO_4 on the P6 watershed is accurate, but it is consistent for different conditions and, therefore, is useful in measuring the size of change in PO_4 washoff for different SWCPs. This is our primary goal in nutrient simulation in this study.

Frequency Analysis

Frequency analysis was used to determine the frequency or probability of occurrence of runoff and pollutant levels on the two watersheds. Analysis was conducted on the interval runoff data produced during the 10-yr simulation periods under a range of meteorologic and environmental conditions. The results of the frequency analysis are compared to determine how these frequencies change with each different SWCP.

P2 Watershed (Georgia) Results--

The frequency curves for the P2 watershed are shown in Figures F-7 through F-11. The curves show the percent of time the particular constituent (for example, runoff in m^3/s) is greater than the ordinate value. For example, Figure F-7 shows that sediment concentrations for the terrace SWCP are greater than 9.0 g/l for 2 percent of the time (with time defined as time during which runoff is occurring and is printed to a computer file). In the same figure it can be seen that the base condition produces sediment concentrations greater than 12.7 mg/l for 2 percent of the time. Each of the SWCP curves for each constituent can be analyzed in this manner.

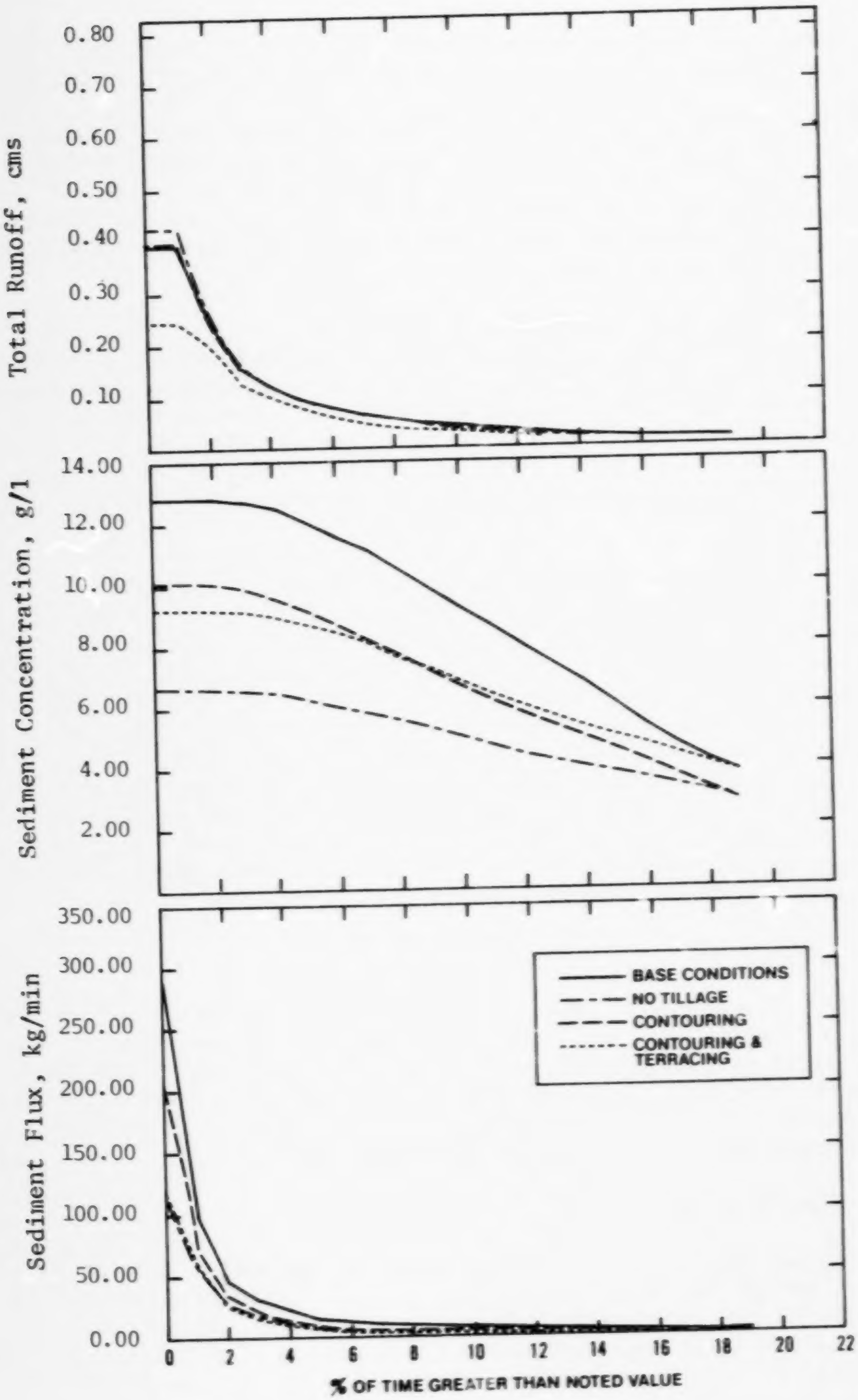

FIGURE F-7. P2 RUNOFF AND SEDIMENT FREQUENCY ANALYSIS

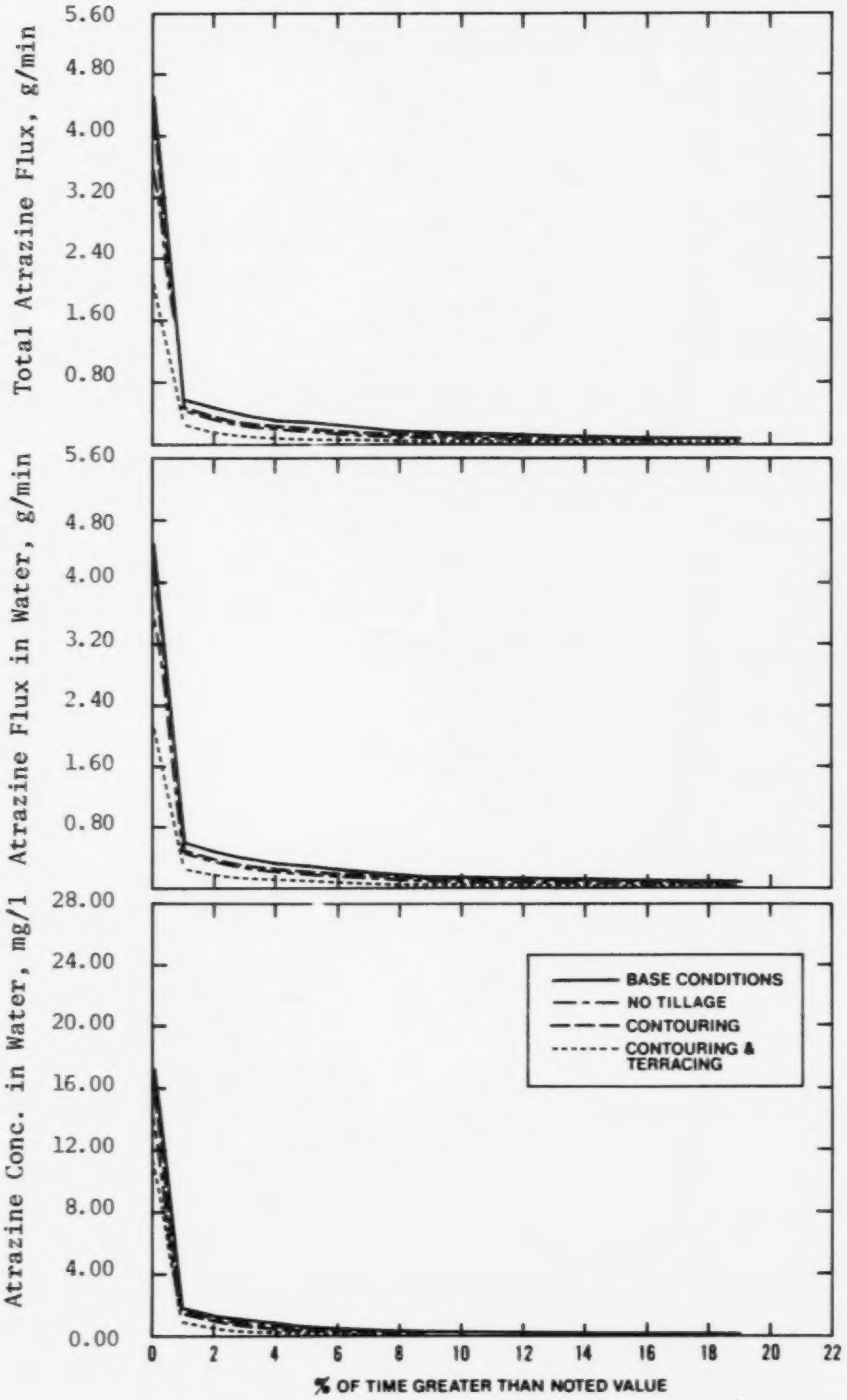

FIGURE F-8. P2 ATRAZINE FREQUENCY ANALYSIS

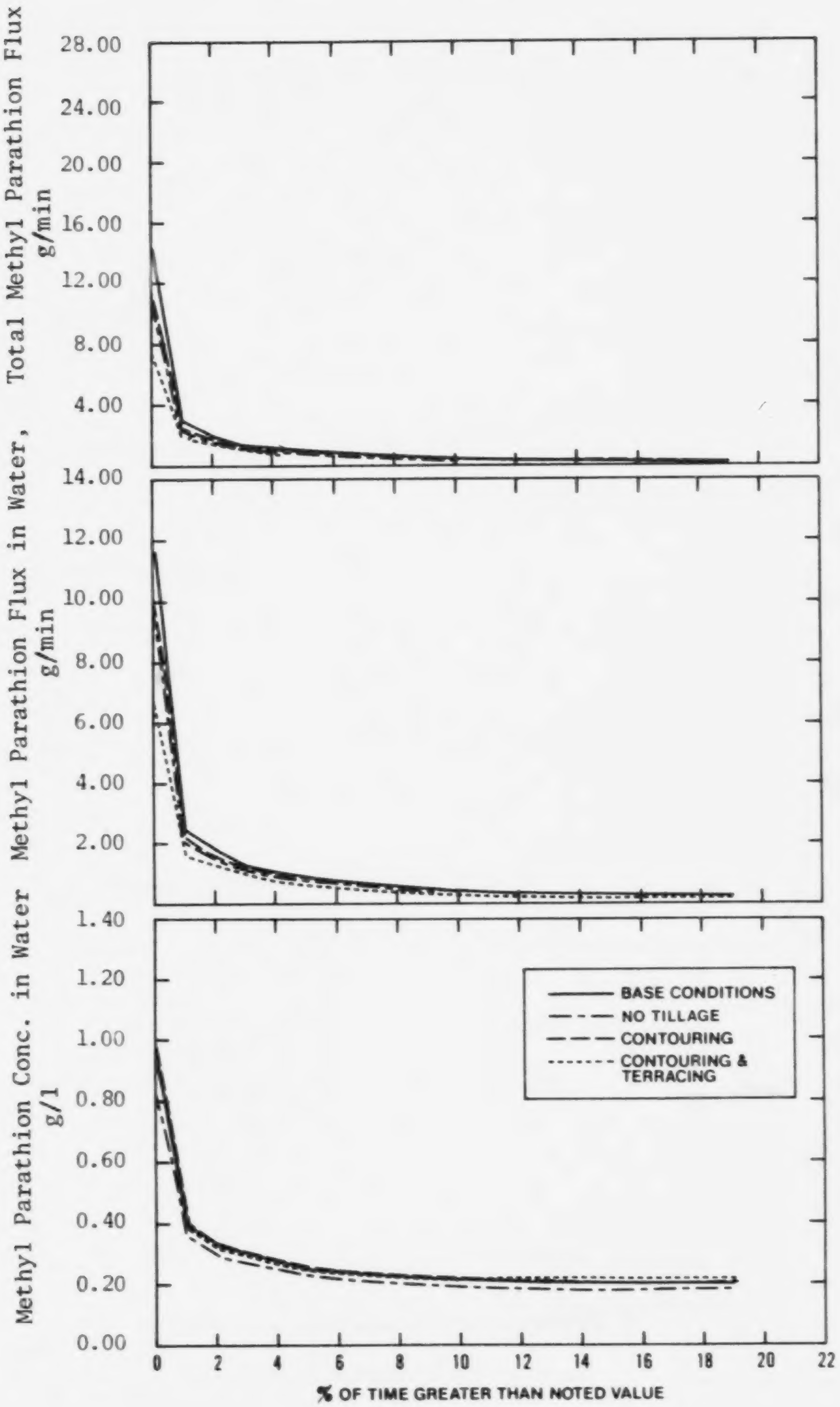

FIGURE F-9. P2 METHYL PARATHION FREQUENCY ANALYSIS

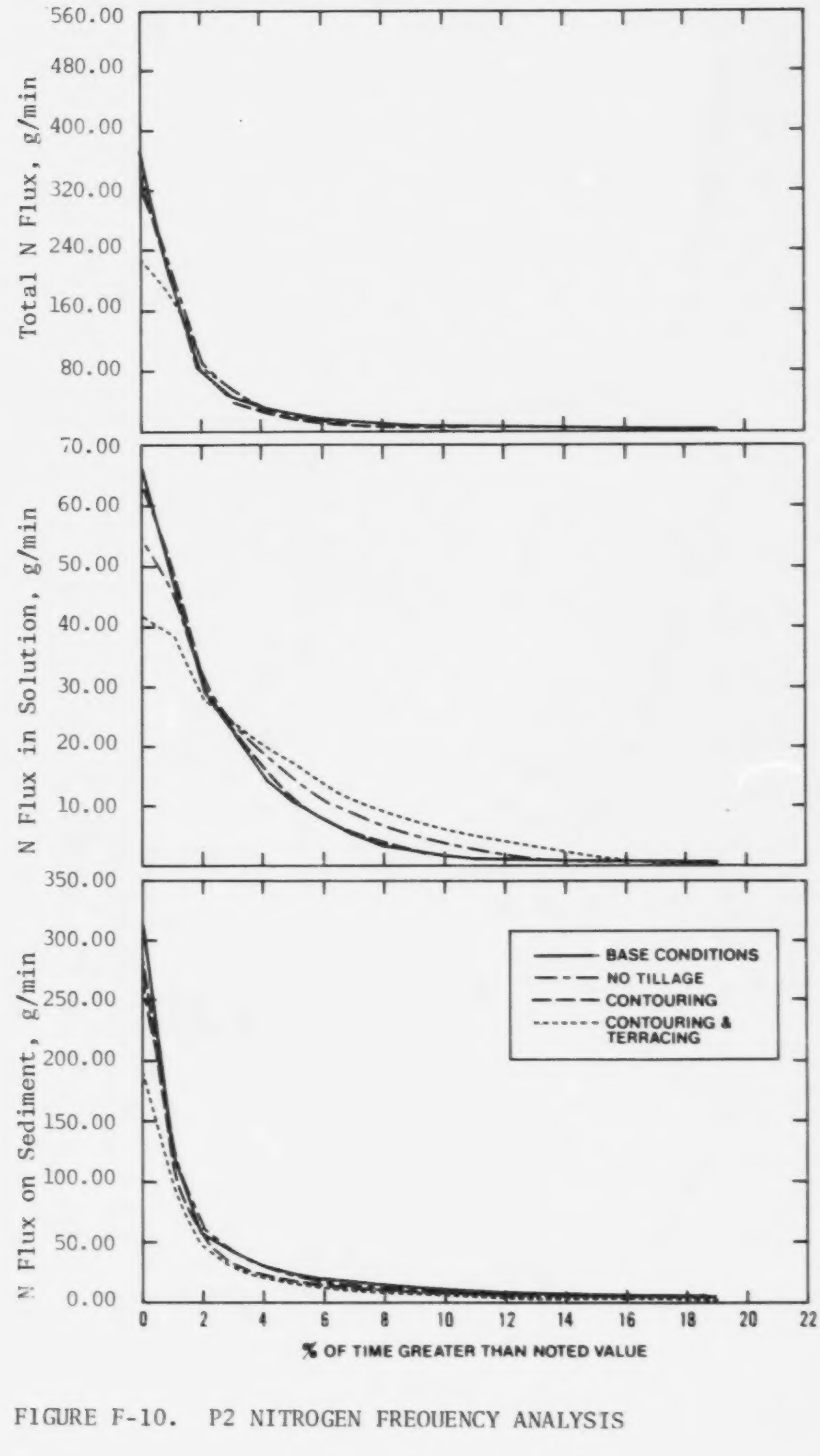

FIGURE F-10. P2 NITROGEN FREQUENCY ANALYSIS

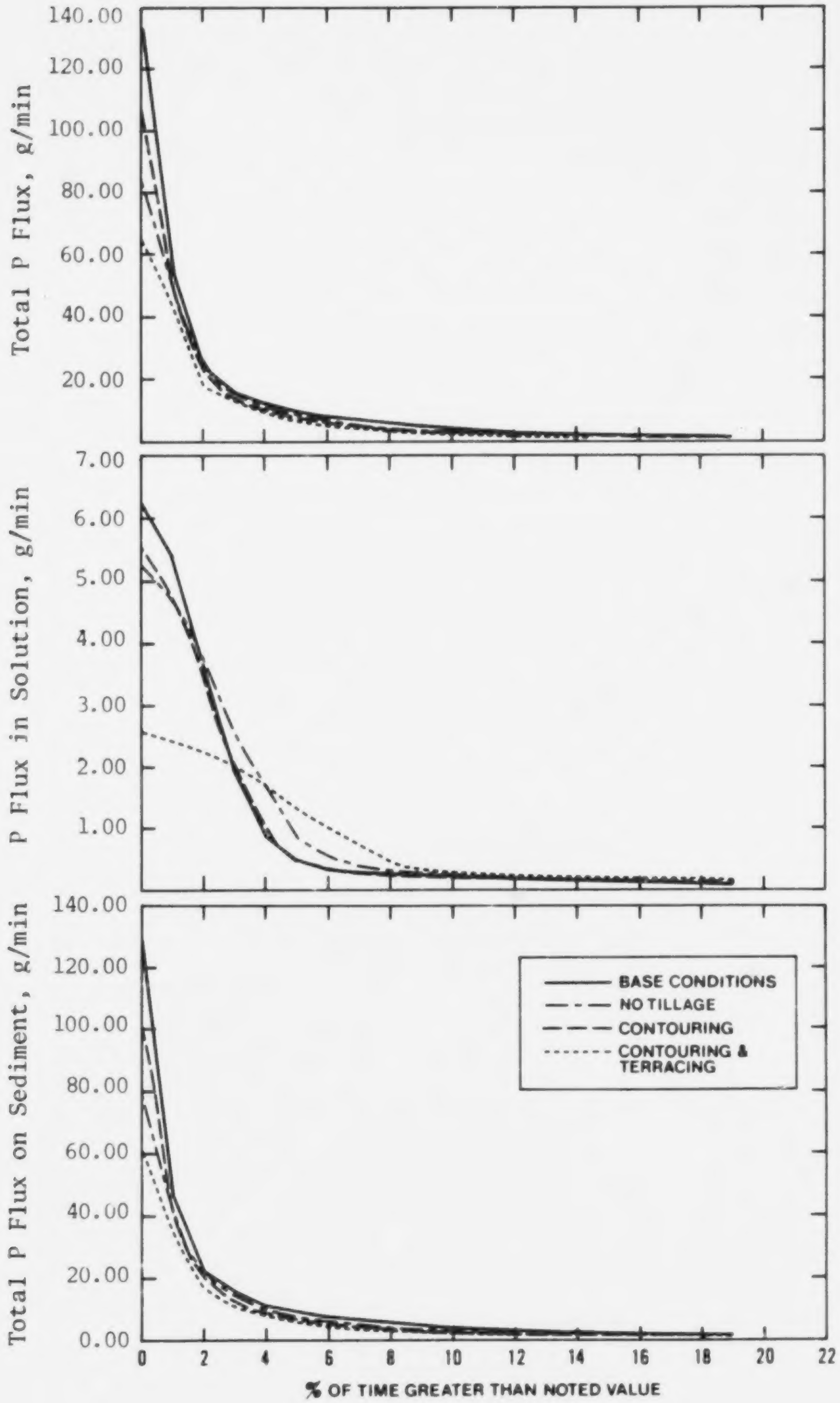

FIGURE F-11. P2 PHOSPHORUS FREQUENCY ANALYSIS

The runoff frequency graph (top graph in Figure F-7) shows relatively small difference among the curves for base conditions, no tillage, and contours, with the combined contours and terraces curve being somewhat lower. The mean annual runoff values in Table F-4 confirm this. The sediment concentration frequency (middle graph in Figure F-7) indicates the base condition concentrations are substantially higher than the three SWCPs which produce similar values. However, the sediment flux frequency (bottom graph in Figure F-4) more closely demonstrates the relative ranking of the SWCPs as shown by the percent reductions in annual values in Table F-4. In general, flux or mass removal is more representative of the effects of a particular SWCP because it reflects changes in both concentration and flow.

Figure F-8 includes the frequency curves for total atrazine flux, atrazine flux in solution, and atrazine concentration in solution. For 1.0 percent of the time the atrazine flux is greater than 0.3 g/min with contours and terraces, and greater than 0.6 g/min for the other watershed conditions. Since very little atrazine is transported on sediment, there is no significant difference between the flux curves for total atrazine and solution atrazine. The concentration curve in Figure F-8 shows that for 1.0 percent of the time, the atrazine concentration in water is greater 1.0 mg/l with contours and terraces, and greater than about 1.5 mg/l for the other conditions. Unfortunately the curve definition in these figures is not sufficient to clarify the differences between the curves. However, with expanded scales the differences would be more readily apparent.

The frequency curves for methyl parathion (Figure F-9) are similar to the atrazine curves. The curves of both pesticides show contours and terraces to produce the largest reduction in the pesticides' flux values. These results are expected. Although the two pesticides have different characteristics, both are primarily removed from the watershed in solution form. Thus, the same runoff results for a particular SWCP will produce washoff of atrazine and methyl parathion in approximately the same proportions, only the ordinate values differ.

The frequency curves can be analyzed to determine how often specific runoff volumes, flow rates, pollutant concentrations, or flux rates will occur. To evaluate potential ecologic impact, the frequency curves and toxicity data can be used to estimate how often acute or chronic pesticide levels toxic to specific organisms will exist.

Paraquat flux and concentration frequency curves are not shown here because paraquat exhibits relatively constant concentrations and, except for scale, the flux curves are identical to the sediment flux curves in Figure F-7.

Total flux frequency curves for nitrogen and phosphorus and their solution and sediment components are shown in Figures F-10 and F-11, respectively. Solution nitrogen includes NO_3 (plus NO_2) and soluble NH_4 while sediment nitrogen is composed of organic nitrogen and adsorbed NH_4. Contours and terraces produce reductions in total and sediment nitrogen flux curves. However, for solution nitrogen the SWCP of combined contours and terraces reduces the high flux rates but increase the rates that occur 96 percent of the time. Also, contours alone increase the solution nitrogen export above the base conditions for values that occur 2 percent of the time or less. Decisions on implementing contours and /or terraces should consider the relative impacts of solution and sediment nitrogen.

The phosphorus flux curves in Figure F-11 show similar results to the nitrogen flux curves. Sediment phosphorus includes organic phosphorus and adsorbed PO_4 while solution phosphorus is comprised only of dissolved PO_4. The effects of contours and terraces on solution phosphorus are similar to those noted above for solution nitrogen. Contours produce lower flux rates while terraces and contours produce a flatter frequency curve with the highest flux rates for a portion of the time.

To evaluate the net or overall impact of the alternative practices (and to quantify the differences between the curves), the area beneath the curve for each practice can be calculated and compared. From elementary decision theory this area represents the expected value of the ordinate variable under all conditions; that is, the value of the variable times its probability of occurrence, summed over all possible occurrences. For example, the area beneath the base sediment curve in Figure F-7 is the measured units of the y-axis, mg/l; each block (1 x-axis unit*1 y-axis unit) is 0.08 mg/l (4 mg/l*.02).

The differences in area beneath each curve, or the area between the curves, can be used to evaluate the impact of a particular practice. Table F-6 lists the area beneath each frequency curve and the percent change for each practice from the base conditions for the P2 watershed. The percent changes are similar to and generally slightly lower than the corresponding values in Table F-4. The difference in the expected value results and the mean annual value results is not usually significant and is due to the decision to only analyze frequency data when the runoff from the watershed is above a minimum value (set to 0.003 m^3/s). This eliminates some small portion of the flow and pollutant load from the frequency analysis that is included in the mean annual values. Frequency analysis provided values for expected pollutant concentrations which are important for analyzing the impact of toxic pollutants, and our analysis procedure eliminates spurious concentrations that may occur at low flows. However, both mean annual values and expected values can mask the effects of extreme conditions. For that reason, visual interpretation of the frequency curves is important to a full understanding of how conditions vary when alternatives are compared.

P6 Watershed (Michigan) Results--

The P6 frequency curves are found in Figures F-12 through F-15. The frequency results are presented in the same format as those for the P2 watershed. The frequency results presented and discussed for the P6 watershed include runoff, sediment, atrazine, nitrogen and phosphorus.

Analysis of the total runoff curves (Figure F-12) shows that for 2 percent of the time that peak runoff is greater for the contour and terrace SWCP than for any other SWCP or the base condition. This is not expected, but can be understood in terms of the flatter slopes of the terrace SWCP not

TABLE F-6. P2 FREQUENCY ANALYSIS OF ALTERNATIVE SWCPs

	Expected Value				Percent Change from Base		
	Base Condition	No Tillage	Contours	Terraces and Contours	No Tillage	Contours	Terraces and Contours
Total Runoff (cm · 10^{-2})	1.94	1.90	1.86	1.43	- 2	- 4	-26
Overland Flow (cm · 10^{-2})	1.87	1.82	1.80	1.32	- 3	- 4	-30
Interflow (cm · 10^{-2})	0.082	0.093	0.085	0.113	+14	+ 3	+38
Sediment Loss							
concentration (g/l)	2.07	1.39	1.56	1.75	-33	-25	-16
flux (kg/min)	4.83	2.42	3.30	2.48	-50	-32	-49
Total Atrazine Flux (g/min)	0.0772	0.0657	0.0642	0.0293	-15	-17	-62
Atrazine Loss in solution							
concentration (mg/l)	0.199	0.169	0.172	0.108	-15	-14	-46
flux (g/min)	0.0764	0.0649	0.0637	0.0290	-15	-17	-62
Atrazine Loss on sediment							
concentration (mg/l)	0.608	0.553	0.536	0.456	- 9	-12	-25
flux (g/min)	0.0008	0.0007	0.0006	0.0004	-12	-25	-50
Paraquat Loss on sediment							
concentration (mg/l)	50.74	55.27	53.85	56.04	+ 9	+ 6	+10
flux (g/min)	0.322	0.168	0.232	0.174	-48	-28	-46
Total Methyl Parathion Flux (g/min)	0.2736	0.2258	0.2353	0.1596	-18	-14	-42
Methyl Parathion Loss in solution							
concentration (mg/l)	0.1281	0.1167	0.1280	0.1268	- 9	+ 0	- 1
flux (g/min)	0.2418	0.2069	0.2178	0.1476	-14	-10	-39
Methyl Parathion Loss on sediment							
concentration (mg/l)	3.3439	3.0886	3.3933	3.6092	- 8	+ 2	+ 8
flux (g/min)	0.0316	0.0192	0.0183	0.0115	-39	-42	-64
Total N Flux (g/min)	8.02	7.66	7.15	6.60	- 4	-11	-18
in solution	1.95	2.17	2.01	2.23	+11	+ 3	+14
on sediment	6.25	5.56	5.19	4.32	-11	-17	-31
Total P Flux (g/min)	2.62	2.02	2.14	1.81	-23	-18	-31
in solution	0.186	0.218	0.182	0.165	+17	- 2	-11
on sediment	2.48	1.82	1.97	1.63	-27	-21	-34

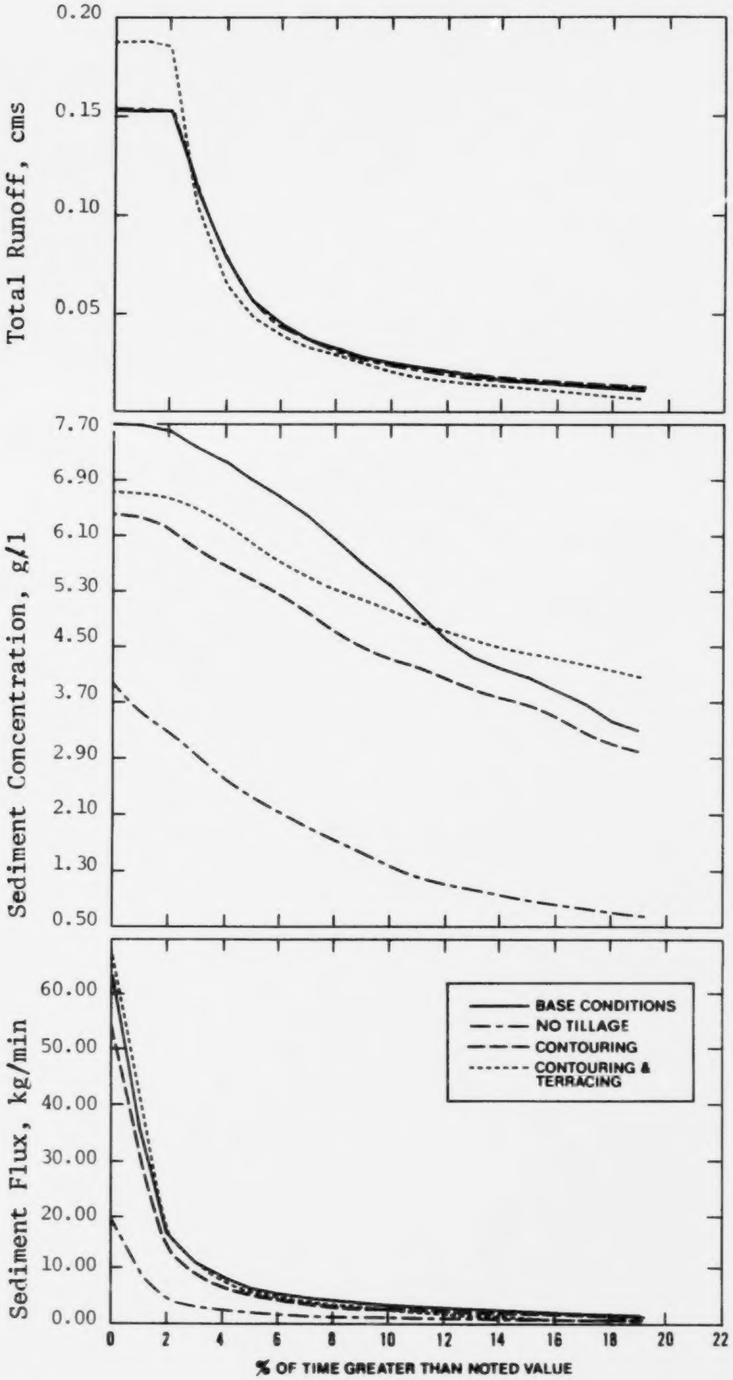

FIGURE F-12. P6 RUNOFF AND SEDIMENT FREQUENCY ANALYSIS

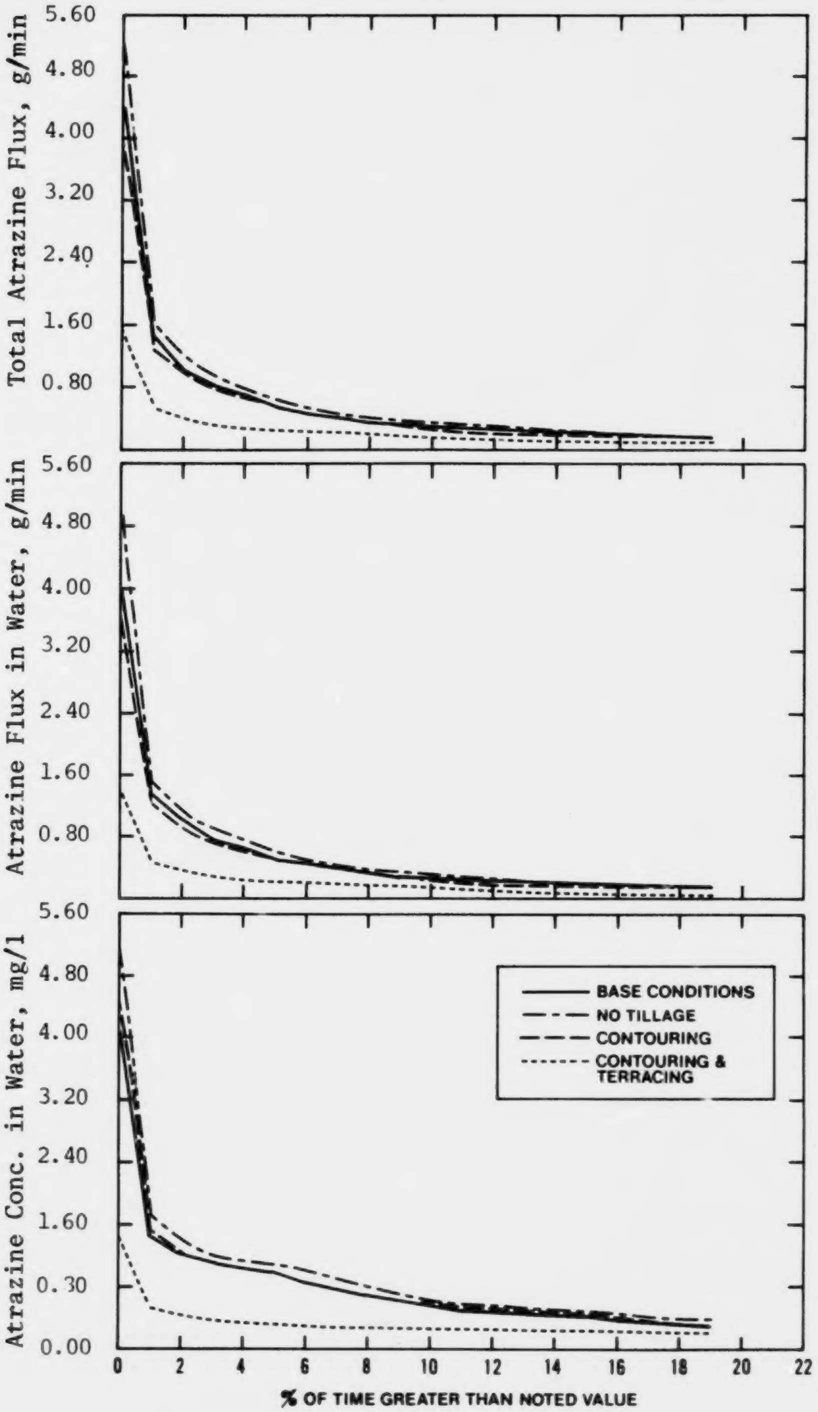

FIGURE F-13. P6 ATRAZINE FREQUENCY ANALYSIS

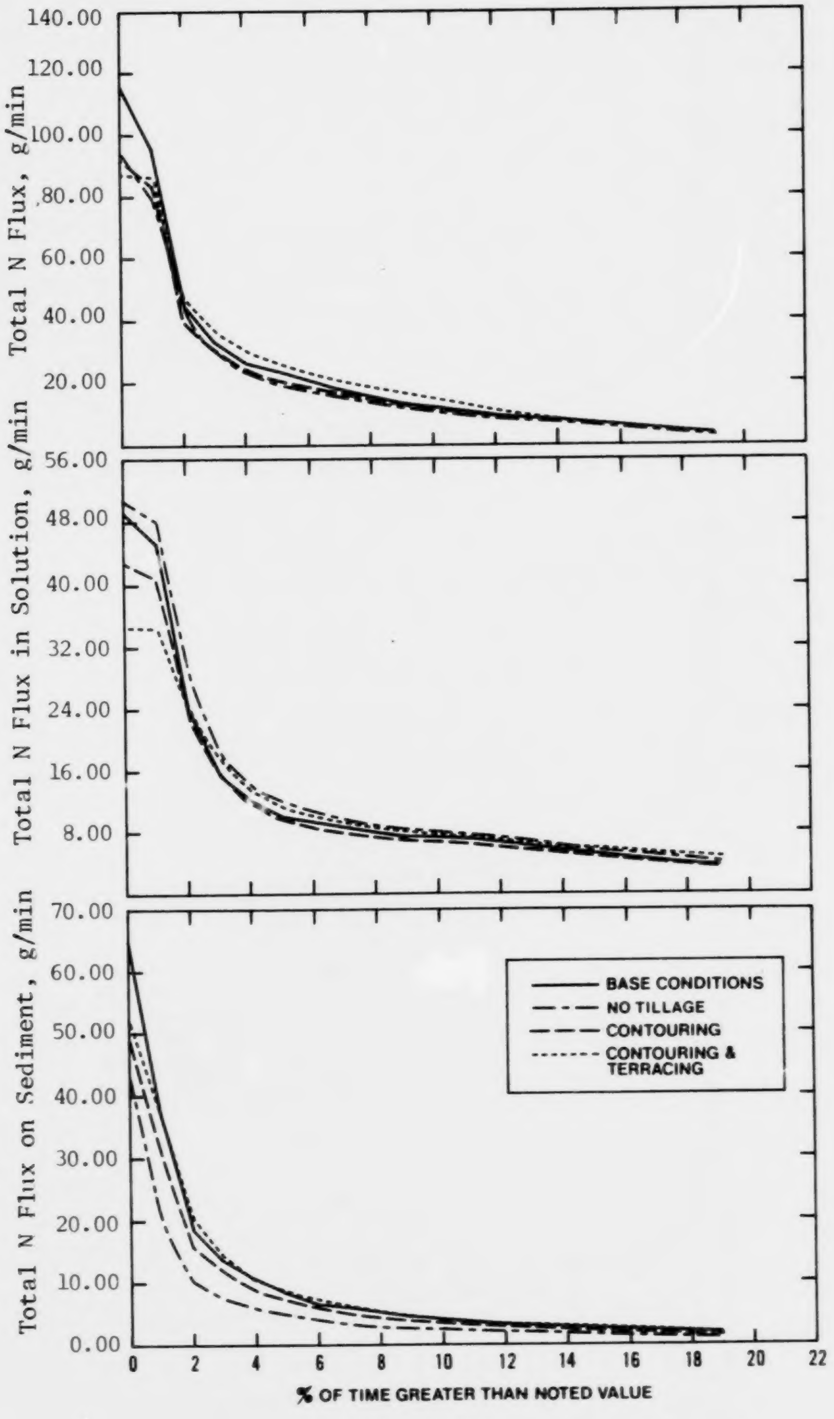

FIGURE F-14. P6 NITROGEN FREQUENCY ANALYSIS

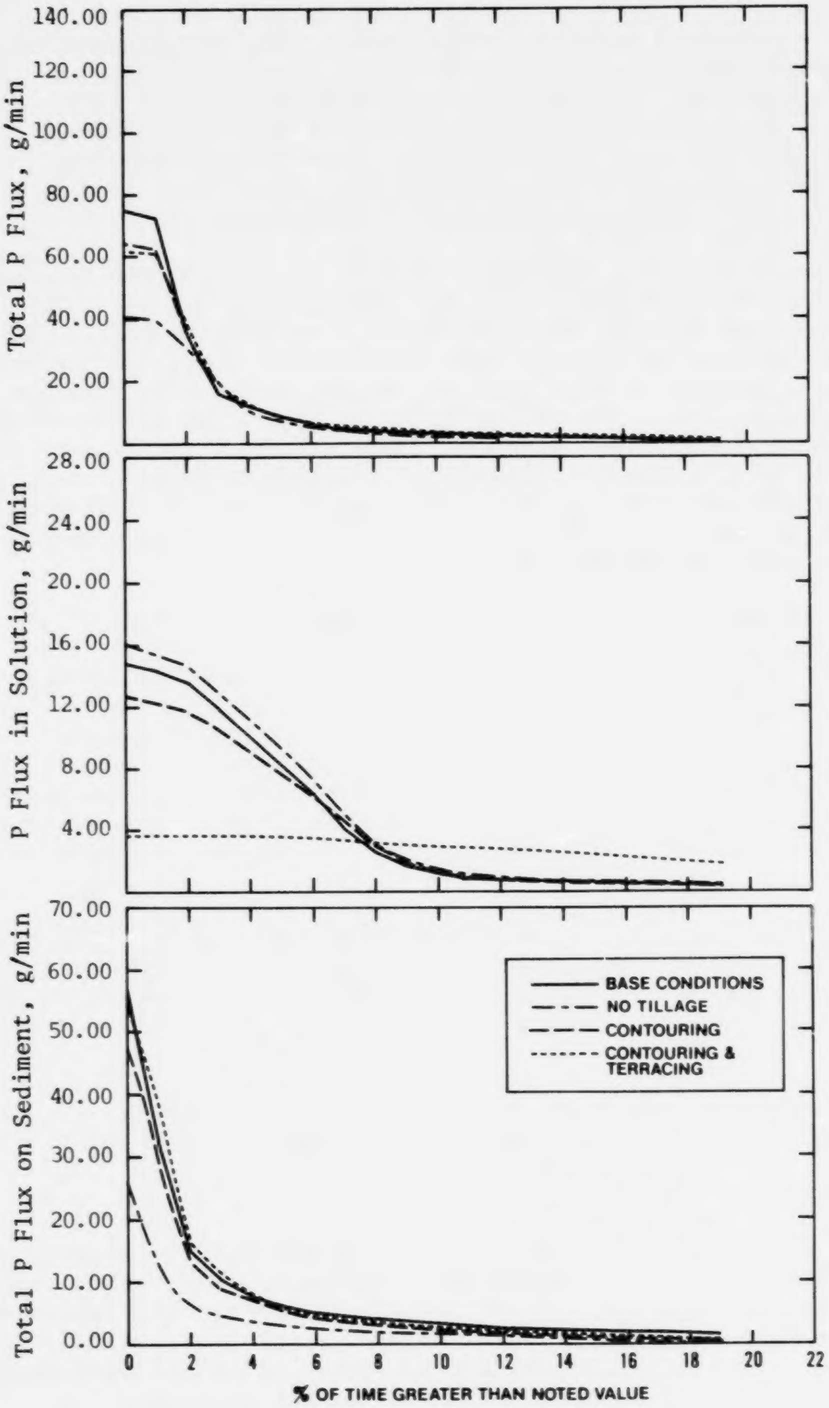

FIGURE F-15. P6 PHOSPHORUS FREQUENCY ANALYSIS draining as quickly after the spring snowmelt as is the case for the other SWCPs and the base conditions. Thus, when a large storm event occurs early in the spring season, higher peak flows are produced by the terraces than under terraces than under other conditions because the terrace soil moisture levels are quite high compared to the other practices. Sediment flux is affected in the same way. No tillage gives the largest reduction in sediment flux and concentration. This reduction in sediment loss is a result of the decrease in soil fines generated due to residue ground cover.

The frequency curves for total atrazine flux, atrazine flux in solution, and atrazine concentration in solution are shown in Figure F-13. Both contours and contours and terraces decrease atrazine washoff as would be expected. However, no tillage increases atrazine loss. The increase is not evident, nor may it be significant as the increase in atrazine flux is less than 10 percent. The increase is most likely the result of one particular storm event which produced a large volume of runoff soon after the application of atrazine in the spring. Whether one wants to consider this increase significant or not, it is an example of how the timing of a meteorological event together with a SWCP can produce the opposite response of what is expected and desired.

The nitrogen results in Figure F-14 show a combination of effects. The terrace SWCP produces the largest total nitrogen flux of the three practices because of the large volume of interflow which transports nitrogen in solution from the watershed. The nitrogen flux in solution is also high for no tillage. This can be related to the large amount of nitrogen left on the watershed after harvest in the form of crop residue. The amount of organic N left as residue is approximately 2.5 times greater than the amount when the residue is removed after harvest (see Table F-2). Much of this organic N is leached from the residue and becomes soluble in the spring snowmelt and washoff.

The phosphorus flux curves (Figure F-15) are similar to the nitrogen flux curves previously discussed. The effect of no tillage on phosphorus flux is less than that observed for nitrogen flux because PO_4 adsorbs onto soil particles, and thus is not as easily removed by subsurface flow, as NO_3.

The area under each curve for the different SWCPs and constituents and the percent change from base conditions are summarized in Table F-7.

CONCLUSIONS

The purpose of this study is to use the ARM Model to analyze the relative effectiveness of alternative SWCPs for controlling runoff, sediment loss, and pesticide and nutrient washoff from small agricultural watersheds. The results of this study provide interesting insight into the influence various SWCPs have in altering the hydrologic response of agricultural areas and how these practices can be represented in a modeling approach.

It has been shown that, in general, the SWCPs studied in this report reduce surface runoff, sediment loss, and associated constituent loads.

TABLE F-7. WATERSHED P6 FREQUENCY ANALYSIS OF ALTERNATIVE SWCPs

	Expected Value				Percent Change from Base		
	Base Condition	No Tillage	Contours	Contours & Terraces	No Tillage	Contours	Contours & Terraces
Total Runoff ($cms \cdot 10^{-2}$)	1.11	1.11	1.10	1.00	+ 0	+ 0	-10
Overland Flow ($cms \cdot 10^{-2}$)	1.05	1.04	1.05	0.91	- 1	+ 0	-13
Interflow ($cms \cdot 10^{-2}$)	0.055	0.065	0.059	0.074	+10	+ 0	+24
Sediment Loss							
concentration (g/1)	1.56	0.50	1.34	1.74	-68	-14	-12
flux (kg/min)	1.74	0.51	1.49	1.56	-71	-14	-10
Total Atrazine Flux (g/min)	0.140	0.147	0.123	0.052	+ 5	-12	-63
Atrazine Loss in solution							
concentration (g/1)	0.220	0.257	0.222	0.104	+17	+ 1	-53
flux (g/min)	0.136	0.145	0.119	0.050	+ 7	-12	-63
Atrazine Loss on sediment							
concentration (g/1)	1.621	1.942	1.666	1.102	+20	+ 3	-32
flux (g/min)	0.0048	0.0032	0.0036	0.0023	-33	-25	-52
Paraquat Loss on sediment							
concentration (g/1)	49.43	54.43	51.00	52.35	+10	+ 3	+ 6
flux (g/min)	0.0995	0.0300	0.0877	0.0930	-70	-12	- 6
Total N Flux (g/min)	4.77	4.20	4.48	4.89	-12	- 6	+ 2
in solution	2.81	3.14	2.66	2.91	+12	- 5	+ 4
on sediment	2.09	1.13	1.89	1.94	-46	-10	- 7
Total P Flux (g/min)	2.46	1.65	2.25	2.22	-33	- 8	-10
in solution	0.897	0.994	0.827	0.707	+11	- 8	-21
on sediment	1.57	0.67	1.43	1.50	-57	- 9	- 4

However, notable exceptions do exist. Practices that reduce and retard overland flow tend to increase subsurface flow components. On both watersheds interflow was increased for each SWCP with the largest increase of 38 percent computed on the P2 watershed for terraces plus contours. This can result in an increase in soluble pollutant washoff. The soluble nitrogen results (mostly NO_3) from the P2 watershed demonstrate this increase. Soluble NH_4 and PO_4 do not exhibit such a strong relationship, but increase for no tillage and decrease for contours with and without terraces. Atrazine and methyl parathion loss occur mostly in solution form and decrease with the different SWCPs in relation to their decrease in total runoff. The adsorption characteristics appear to minimize any increase in the removal of soluble pollutants. This is not true for non-adsorbing constituents such as NO_3, which can substantially increase with subsurface flow.

Sediment and associated constituents decrease for all SWCPs studied. The percent changes for sediment-associated pollutants are generally far larger than for water-related pollutants. The difference can be explained by the fact that the original intent of the three SWCPs studied (no tillage, contours, and contours with terraces) has been to reduce erosion and soil loss from croplands. All three SWCPs appear to be well designed for this purpose.

In this study, we found that terraces with contours is the most effective SWCP for most constituents on both watersheds studied. The contour SWCP and the no tillage SWCP can be considered equally effective on the two watersheds. No tillage is generally better than contours in reducing sediment-associated pollutants. Contours was found to be more effective in decreasing soluble constituents. However, the individual practices defined by the SWCPs vary across the United States, and only two regions are represented in this study. In actual practice various combinations of SWCPs are often used and the specific names are applied to widely differing practices across the country.

No detailed attempt has been made to compare the SWCP results for the two watersheds simulated in this study. However, with one watershed (P2) representing the Piedmont region of the southeastern United States and the other (P6) located in the Great Lakes Basin some obvious differences can be found. The major hydrologic and climatic difference between the two watersheds is that P6 is snow covered much of the winter. P2 rarely receives snow. This difference is important in evaluating the effectiveness of the SWCPs. Snowfall is a much less intense form of precipitation than rainfall, especially thunderstorm rainfall commonly produced in the southeast. Thus, for the months of the year when snow provides a protective manner, regions of the country which rarely have intensive rainfall, such as the Pacific Northwest, will not benefit as much from SWCPs as areas subject to heavy thunderstorm activity.

Parameter changes in the ARM Model to represent the alternative SWCPs were based on information in the literature and our own judgment and experience in applying the ARM Model on other watersheds. When little or no information was available, we tended to be conservative in the parameter adjustments so as not to overstate the impacts of the different practices.

Also, limitations within the model and lack of data prevented consideration of other effects such as changes in infiltration characteristics, crop growth, nutrient transformation, etc. Research is needed to define relationships between model parameters and measurable watershed characteristics, and additional data collection is needed to characterize the impacts of various SWCPs as a basis for parameter adjustments.

Although the ARM Model is by no means exact in its representation of all soil processes, this work has shown that the model can be used to analyze the relative effects of some alternative SWCPs. Further research is needed to better represent erosion processes, the effects of tillage operations, the transport of soluble substances, pesticide adsorption and degradation mechanisms, and nutrient transformations. However, for most processes the model includes our best current understanding of the important mechanisms. With an awareness of the changes affected by a SWCP and the representation of these changes by the model the important effects of a SWCP can be investigated.

ACKNOWLEDGEMENTS

This study and report were funded by Cornell University in conjunction with a research grant from the U.S. Environmental Protection Agency. Dr. Raymond C. Loehr and Dr. Douglas A. Haith and their staff at the Department of Agricultural Engineering at Cornell University provided ideas, suggestions, and comments on our work which were invaluable to the successful completion of this study. In addition, Mr. Lee Mulkey and Mr. Charles Smith of the Environmental Research Laboratory in Athens, Georgia were very helpful in reviewing some of the ideas and assumptions used in the modeling of methyl parathion on the P2 watershed and our concepts of how to represent base conditions for that watershed.

At Hydrocomp, Mr. Douglas Beyerlein was project manager and was responsible for the use of the ARM Model for simulation of soil and water conservation practices and the content of this report. Mr. Anthony S. Donigian and Dr. Norman Crawford provided technical guidance and together with Mr. Beyerlein represented Hydrocomp, Inc. at Cornell's Panel of Experts discussions. Mr. Harley Davis, Jr. provided the input for and helped review the nutrient simulation. Mr. Kevin Gartner assembled the meteorologic data required for the 10-yr simulations and conducted the mean annual value and frequency analysis. Mr. Tamas Eger and Mr. Gartner are responsible for the computer programs which produced these analyses from the ARM Model output. The production of this report would not be possible without the able editing of Ms. Donna Mitchell, the swift typing of Ms. Kathy Francies, and the clear graphics of Mr. Guy Funabiki.

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P2 and P6 PARAMETER VALUES

The P2 and P6 parameter values used in the ARM Model are presented in the form of input sequences to the model. With each major pollutant simulated (atrazine, paraquat, methyl parathion, and nutrients) are the input sequences for the four practices simulated (base condition, minimum tillage, contours, and terraces and contours).

P2 Parameter Values

Atrazine--

```
P-2 WATKINSVILLE, GA.: BASE CONDITIONS (NO SWCP)
ATRAZINE APPLIED: 2.6 LBS/AC EVERY DAY 129 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.42 UZS=    0.420 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.2000 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=     0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.100 0.100 0.100 0.100 0.000 0.150 0.600 0.850 0.750 0.100 0.100 0.100
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.600 SRERI= 2.00 SCMPAC 0.02
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  129 129 129 129 129 129 129 129 129 129 000 000 NPA= 10 NDRPA=  2
 YEARAP=  66  67  68  69  70  71  72  73  74  75  99  99
 SSTR=   2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 0.000 0.000
 CMAX=  3.5E-5 DD=    0.0000 K=      2.000 N=      1.0000 NP=    2.300
 DDG=    129 138 129 138 129 138 129 138 129 138 129 138
 YDG=     66  66  67  67  68  68  69  69  70  70  71  71
 KDG=    0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

```
P-2 WATKINSVILLE, GA.: SWCP -- NO TILLAGE
ATRAZINE APPLIED: 2.6 LBS/AC EVERY DAY 129 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=     3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.42 UZS=     0.420 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.3200 A=    0.0000 EPXM=0.1200 PETML 1.000
 K3=      0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.600 0.600 0.600 0.600 0.400 0.490 0.760 0.910 0.850 0.600 0.600 0.600
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.550 SRERI= 0.20 SCMPAC 0.02
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  129 129 129 129 129 129 129 129 129 129 000 000 NPA= 10 NDRPA=  2
 YEARAP=  66  67  68  69  70  71  72  73  74  75  99  99
 SSTR=   2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 0.000 0.000
 CMAX=  3.5E-5 DD=     0.0000 K=       2.000 N=      1.0000 NP=     2.300
 DDG=     129 138 129 138 129 138 129 138 129 138 129 138
 YDG=      66  66  67  67  68  68  69  69  70  70  71  71
 KDG=     0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

```
P-2 WATKINSVILLE, GA.: SWCP -- CONTOURING
ATRAZINE APPLIED: 2.6 LBS/AC EVERY DAY 129 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.50 UZS=    0.500 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=     0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.100 0.100 0.100 0.100 0.000 0.150 0.600 0.850 0.750 0.100 0.100 0.100
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=0.5
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.500 SRERI= 2.00 SCMPAC 0.02
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  129 129 129 129 129 129 129 129 129 129 000 000 NPA= 10 NDRPA=  2
 YEARAP=  66  67  68  69  70  71  72  73  74  75  99  99
 SSTR=   2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 0.000 0.000
 CMAX=  3.5E-5 DD=    0.0000 K=      2.000 N=     1.0000 NP=    2.300
 DDG=    129 138 129 138 129 138 129 138 129 138 129 138
 YDG=     66  66  67  67  68  68  69  69  70  70  71  71
 KDG=    0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

```
P-2 WATKINSVILLE, GA.: SWCP - TERRACES (2) WITH CONTOURING
ATRAZINE APPLIED: 2.6 LBS/AC EVERY DAY 129 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.65 UZS=    0.650 LZSN=   18.00 LZS=    18.00
 L=     250. SS=  0.0150 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=     0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.100 0.100 0.100 0.100 0.000 0.150 0.600 0.850 0.750 0.100 0.100 0.100
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=0.35
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.500 SRERI= 2.00 SCMPAC 0.02
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  129 129 129 129 129 129 129 129 129 129 000 000 NPA= 10 NDRPA=  2
 YEARAP=  66  67  68  69  70  71  72  73  74  75  99  99
 SSTR=   2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 2.600 0.000 0.000
 CMAX=  3.5E-5 DD=     0.0000 K=        2.000 N=       1.0000 NP=     2.300
 DDG=    129 138 129 138 129 138 129 138 129 138 129 138
 YDG=     66  66  67  67  68  68  69  69  70  70  71  71
 KDG=    0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040 0.100 0.040
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

Paraquat--

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P-2 WATKINSVILLE, GA.: BASE CONDITIONS (NO SWCP)
PARAQUAT APPLIED: 2.25 LBS/AC EVERY DAY 129 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.42 UZS=    0.420 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.2000 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=      0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.100 0.100 0.100 0.100 0.000 0.150 0.600 0.850 0.750 0.100 0.100 0.100
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.600 SRERI= 2.00 SCMPAC 0.02
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  129 129 129 129 129 129 129 129 129 129 000 000 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  99  99
 SSTR=   2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 0.000 0.000
 CMAX=  1.0E-5 DD=    0.0003 K=       120.0 N=     2.0000 NP=    4.600
 DDG=    129
 YDG=     66
 KDG=    0.002
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

```
P-2 WATKINSVILLE, GA.: SWCP -- NO TILLAGE
PARAQUAT APPLIED: 2.25 LBS/AC EVERY DAY 129 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.42 UZS=    0.420 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.3200 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=      0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.600 0.600 0.600 0.600 0.400 0.490 0.760 0.910 0.850 0.600 0.600 0.600
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.550 SRERI= 0.20 SCMPAC 0.02
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  129 129 129 129 129 129 129 129 129 129 000 000 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  99  99
 SSTR=   2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 0.000 0.000
 CMAX=  1.0E-5 DD=    0.0003 K=      120.0 N=     2.0000 NP=    4.600
 DDG=    129
 YDG=     66
 KDG=    0.002
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

```
P-2 WATKINSVILLE, GA.: SWCP — CONTOURING
PARAQUAT APPLIED: 2.25 LBS/AC EVERY DAY 129 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.50 UZS=    0.500 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=      0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=    0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.100 0.100 0.100 0.100 0.000 0.150 0.600 0.850 0.750 0.100 0.100 0.100
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=0.5
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.500 SRERI= 2.00 SCMPAC 0.02
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  129 129 129 129 129 129 129 129 129 129 000 000 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  99  99
 SSTR=   2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 0.000 0.000
 CMAX=  1.0E-5 DD=    0.0003 K=       120.0 N=      2.0000 NP=    4.600
 DDG=    129
 YDG=     66
 KDG=    0.002
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

```
P-2 WATKINSVILLE, GA.: SWCP - TERRACES (2) WITH CONTOURING
PARAQUAT APPLIED: 2.25 LBS/AC EVERY DAY 129 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.65 UZS=    0.650 LZSN=   18.00 LZS=    18.00
 L=     250. SS=  0.0150 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=      0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.100 0.100 0.100 0.100 0.000 0.150 0.600 0.850 0.750 0.100 0.100 0.100
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=0.35
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.500 SRERI= 2.00 SCMPAC 0.02
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  129 129 129 129 129 129 129 129 129 129 000 000 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  99  99
 SSTR=   2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 2.250 0.000 0.000
 CMAX=  1.0E-5 DD=     0.0003 K=       120.0 N=      2.0000 NP=    4.600
 DDG=    129
 YDG=     66
 KDG=    0.002
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

Methyl Parathion--

```
P-2 WATKINSVILLE, GA.: BASE CONDITIONS (NO SWCP)
METHYL PARATHION APPLIED: 1.0 LB/AC 6 TIMES PER YR FOR 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.42 UZS=    0.420 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.2000 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=     0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.200 0.200 0.200 0.200 0.000 0.100 0.500 0.750 0.650 0.550 0.200 0.200
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.600 SRERI= 2.00 SCMPAC 0.02
PESTICIDE
APMODE=CROP
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  166 180 194 208 222 236 166 180 194 208 222 236 NPA= 60 NDRPA=  1
 YEARAP=  66  66  66  66  66  66  67  67  67  67  67  67
 SSTR=   1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000
 CMAX=  4.0E-5 DD=    0.0000 K=      25.000 N=      1.0000 NP=    1.000
 PSCZ=  0.0   PCK=    1.16  KCDG=    0.087
 DDG=    166 166 166 166 166 166 166 166 166 166 166 166
 YDG=     66  67  68  69  70  71  72  73  74  75  00  00
 KDG=    .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

```
P-2 WATKINSVILLE, GA.: SWCP -- NO TILLAGE
METHYL PARATHION APPLIED: 1.0 LB/AC 6 TIMES PER YR FOR 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=     3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.42 UZS=     0.420 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.3200 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=      0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.200 0.200 0.200 0.200 0.200 0.280 0.600 0.800 0.720 0.640 0.200 0.200
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.550 SRERI= 0.20 SCMPAC 0.02
PESTICIDE
APMODE=CROP
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  166 180 194 208 222 236 166 180 194 208 222 236 NPA= 60 NDRPA=  1
 YEARAP=  66  66  66  66  66  66  67  67  67  67  67  67
 SSTR=   1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000
 CMAX=  4.0E-5 DD=    0.0000 K=     25.000 N=     1.0000 NP=    1.000
 PSCZ=  0.0   PCK=    1.16  KCDG=   0.087
 DDG=     166 166 166 166 166 166 166 166 166 166 166 166
 YDG=      66  67  68  69  70  71  72  73  74  75  00  00
 KDG=     .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

```
P-2 WATKINSVILLE, GA.: SWCP -- CONTOURING
METHYL PARATHION APPLIED: 1.0 LB/AC 6 TIMES PER YR FOR 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.50 UZS=    0.500 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=      0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.200 0.200 0.200 0.200 0.000 0.100 0.500 0.750 0.650 0.550 0.200 0.200
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=0.5
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.500 SRERI= 2.00 SCMPAC 0.02
PESTICIDE
APMODE=CROP
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  166 180 194 208 222 236 166 180 194 208 222 236 NPA= 60 NDRPA=  1
 YEARAP=  66  66  66  66  66  66  67  67  67  67  67  67
 SSTR=   1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000
 CMAX=  4.0E-5 DD=    0.0000 K=      25.000 N=      1.0000 NP=     1.000
 PSCZ=  0.0   PCK=     1.16  KCDG=    0.087
 DDG=     166 166 166 166 166 166 166 166 166 166 166 166
 YDG=      66  67  68  69  70  71  72  73  74  75  00  00
 KDG=     .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

```
P-2 WATKINSVILLE, GA.: SWCP - TERRACES (2) WITH CONTOURING
METHYL PARATHION APPLIED: 1.0 LB/AC 6 TIMES PER YR FOR 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.65 UZS=    0.650 LZSN=   18.00 LZS=    18.00
 L=     250. SS=  0.0150 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=     0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.200 0.200 0.200 0.200 0.000 0.100 0.500 0.750 0.650 0.550 0.200 0.200
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=0.35
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.500 SRERI= 2.00 SCMPAC 0.02
PESTICIDE
APMODE=CROP
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  166 180 194 208 222 236 166 180 194 208 222 236 NPA= 60 NDRPA=  1
 YEARAP=  66  66  66  66  66  66  67  67  67  67  67  67
 SSTR=   1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000
 CMAX=  4.0E-5 DD=     0.0000 K=      25.000 N=      1.0000 NP=    1.000
 PSCZ=  0.0    PCK=     1.16  KCDG=    0.087
 DDG=     166 166 166 166 166 166 166 166 166 166 166 166
 YDG=      66  67  68  69  70  71  72  73  74  75  00  00
 KDG=     .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154 .0154
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ= 99.9 UZF=3.0 LZF=1.5
```

Nutrients--

```
P-2 WATKINSVILLE, GA.: BASE CONDITIONS (NO SWCP)
NUTRIENTS APPLIED: N & P ON DAY 119; N ON DAY 162; PLANT DECAY AFTER HARVEST
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=NO
NUTR=YES
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=     3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=    0.42 UZS=     0.420 LZSN=    18.00 LZS=     18.00
 L=      100. SS=  0.0250 NN=  0.2000 A=    0.0000 EPXM=0.1200 PETML 1.000
 K3=      0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.100 0.100 0.100 0.100 0.000 0.150 0.600 0.850 0.750 0.100 0.100 0.100
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.600 SRERI= 2.00 SCMPAC 0.02
NUTRIENT
  TSTEP= 360  NAPPL=   3 TIMHAR= 297
 ULUPTK=0.00  0.05  0.08  0.10  0.15  0.75  0.200  0.07 0.00  0.00  0.00   0.0
 LZUPTK=0.00  0.00  0.00  0.05  0.03  0.55  0.450  0.10 0.00  0.00  0.00   0.0

REACTION RATES
NITROGEN
SURFACE
 1.00    0.0     0.10     0.00   0.0     0.0      1.0    .20
UPPER ZONE
 0.20    0.006    0.13    0.002   0.00     0.0     1.0    0.25
LOWER ZONE
 0.10    0.002    0.025    0.002   0.0      0.0      1.0    0.2
GROUNDWATER
 0.0     0.0     0.0     0.0     0.0     0.0      0.0     0.0
TEMPERATURE COEFFICIENTS
 1.05    1.07    1.07    1.07    1.07    1.07    1.05    1.05

PHOSPHORUS
SURFACE
 0.02     0.0     0.01    1.0     0.015
UPPER ZONE
 0.002   0.0     0.70    1.0     0.0015
LOWER ZONE
 0.002    0.0    0.80    1.0     0.005
GROUNDWATER
 0.0     0.0     0.0     0.0     0.0
TEMPERATURE COEFFICIENTS
 1.07    1.07    1.07    1.05    1.05
END
```

```
INITIAL
NITROGEN
SURFACE
 90.0      2.0      20.0     0.20     0.0      0.0
UPPER ZONE
 375.0     4.0      40.0      3.00    0.0      0.0
LOWER ZONE
2500.      6.0     20.0      212.     0.0      0.0
GROUNDWATER
  0.0      0.0      0.0      0.0      0.0      0.0

PHOSPHORUS
SURFACE
  12.0     1.0      36.0     0.0
UPPER ZONE
 100.    0.35     305.0     0.0
LOWER ZONE
 800.      1.0    135.0     0.0
GROUNDWATER
  0.0      0.0      0.0      0.0

CHLORIDE
SURFACE
  3.46
UPPER ZONE
  79.54
LOWER ZONE
  770.
GROUNDWATER
  0.0
END


APPLICATION  119
NITROGEN
SURFACE
     0.0    0.71    0.0       0.0       0.0      0.0
UPPER ZONE
     0.0   33.29     0.0      0.0       0.0      0.0

PHOSPHORUS
SURFACE
     0.0      0.2      0.40    0.0
UPPER ZONE
    0.0      28.9    0.0     0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END
```

```
APPLICATION  162
NITROGEN
SURFACE
    0.0     25.0    3.0      0.0      0.0      0.0
UPPER ZONE
   0.0     61.9    0.0      0.0      0.0      0.0

PHOSPHORUS
SURFACE
     0.0     0.0     0.0     0.0
UPPER ZONE
     0.0     0.0     0.0     0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END


APPLICATION  297
NITROGEN
SURFACE
    1.5     0.0     0.0      0.0      0.0      0.0
UPPER ZONE
  20.0      0.0    0.0      0.0      0.0      0.0

PHOSPHORUS
SURFACE
     0.3     0.0     0.0     0.0
UPPER ZONE
     4.0     0.0     0.0     0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END
 LZTEMP=  48.0  47.0  53.0  57.0  65.0  73.0  77.0  77.0  76.0  71.0  58.0  51.0
 ASZT=  -21.4  BSZT=   1.345 AUZT=  26.210 BUZT=   0.675
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ=99.90 UZF=5.0 LZF=1.0
```

```
P-2 WATKINSVILLE, GA.: SWCP -- NO TILLAGE
NUTRIENTS APPLIED: N & P ON DAY 119; N ON DAY 162; PLANT DECAY AFTER HARVEST
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=NO
NUTR=YES
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.42 UZS=    0.420 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.3200 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=     0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.600 0.600 0.600 0.600 0.400 0.490 0.760 0.910 0.850 0.600 0.600 0.600
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.150 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.550 SRERI= 0.20 SCMPAC 0.02
NUTRIENT
  TSTEP= 360  NAPPL=   3 TIMHAR= 297
 ULUPTK=0.00  0.05  0.08  0.10  0.15  0.75  0.200  0.07 0.00  0.00  0.00   0.0
 LZUPTK=0.00  0.00  0.00  0.05  0.03  0.55  0.450  0.10 0.00  0.00  0.00   0.0

REACTION RATES
NITROGEN
SURFACE
 1.00     0.0      0.10      0.00    0.0      0.0       1.0     .20
UPPER ZONE
 0.20     0.006     0.13     0.002    0.00     0.0      1.0     0.25
LOWER ZONE
 0.10     0.002     0.025     0.002   0.0      0.0       1.0     0.2
GROUNDWATER
 0.0      0.0      0.0      0.0      0.0      0.0       0.0     0.0
TEMPERATURE COEFFICIENTS
 1.05     1.07     1.07     1.07     1.07     1.07     1.05     1.05

PHOSPHORUS
SURFACE
 0.02      0.0      0.01     1.0     0.015
UPPER ZONE
 0.002    0.0      0.70     1.0     0.0015
LOWER ZONE
 0.002     0.0     0.80     1.0     0.005
GROUNDWATER
 0.0      0.0      0.0      0.0     0.0
TEMPERATURE COEFFICIENTS
 1.07     1.07     1.07     1.05     1.05
END
```

```
INITIAL
NITROGEN
SURFACE
 90.0     2.0     20.0    0.20    0.0     0.0
UPPER ZONE
 375.0    4.0     40.0     3.00   0.0     0.0
LOWER ZONE
2500.     6.0    20.0     212.    0.0     0.0
GROUNDWATER
  0.0     0.0     0.0     0.0     0.0     0.0

PHOSPHORUS
SURFACE
  12.0     1.0    36.0    0.0
UPPER ZONE
 100.    0.35    305.0    0.0
LOWER ZONE
 800.     1.0    135.0    0.0
GROUNDWATER
  0.0     0.0     0.0     0.0

CHLORIDE
SURFACE
  3.46
UPPER ZONE
  79.54
LOWER ZONE
  770.
GROUNDWATER
  0.0
END


APPLICATION  119
NITROGEN
SURFACE
     0.0    0.71    0.0       0.0     0.0     0.0
UPPER ZONE
     0.0   33.29     0.0      0.0     0.0     0.0

PHOSPHORUS
SURFACE
     0.0     0.2     0.40    0.0
UPPER ZONE
    0.0     28.9    0.0     0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END
```

```
APPLICATION  162
NITROGEN
SURFACE
    0.0      25.0    3.0      0.0      0.0      0.0
UPPER ZONE
   0.0     61.9    0.0      0.0      0.0      0.0

PHOSPHORUS
SURFACE
     0.0     0.0      0.0      0.0
UPPER ZONE
     0.0     0.0      0.0      0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END


APPLICATION  297
NITROGEN
SURFACE
    8.0     0.0      0.0       0.0      0.0      0.0
UPPER ZONE
  20.0       0.0    0.0      0.0      0.0      0.0

PHOSPHORUS
SURFACE
     1.5     0.0      0.0      0.0
UPPER ZONE
     4.0     0.0      0.0      0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END
 LZTEMP=  48.0  47.0  53.0  57.0  65.0  73.0  77.0  77.0  76.0  71.0  58.0  51.0
 ASZT=  -21.4  BSZT=   1.345 AUZT=  26.210 BUZT=   0.675
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ=99.90 UZF=5.0 LZF=1.0
```

```
P-2 WATKINSVILLE, GA.: SWCP -- CONTOURING
NUTRIENTS APPLIED: N & P ON DAY 119; N ON DAY 162; PLANT DECAY AFTER HARVEST
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=NO
NUTR=YES
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.50 UZS=    0.500 LZSN=   18.00 LZS=    18.00
 L=     100. SS=  0.0250 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 1.000
 K3=     0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.100 0.100 0.100 0.100 0.000 0.150 0.600 0.850 0.750 0.100 0.100 0.100
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=0.5
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.500 SRERI= 2.00 SCMPAC 0.02
NUTRIENT
  TSTEP= 360  NAPPL=   3 TIMHAR= 297
 ULUPTK=0.00  0.05  0.08  0.10  0.15  0.75  0.200  0.07 0.00  0.00  0.00   0.0
 LZUPTK=0.00  0.00  0.00  0.05  0.03  0.55  0.450  0.10 0.00  0.00  0.00   0.0

REACTION RATES
NITROGEN
SURFACE
 1.00    0.0     0.10     0.00   0.0     0.0      1.0    .20
UPPER ZONE
 0.20    0.006    0.13    0.002   0.00     0.0     1.0     0.25
LOWER ZONE
 0.10    0.002    0.025    0.002   0.0     0.0      1.0     0.2
GROUNDWATER
 0.0     0.0     0.0      0.0     0.0     0.0      0.0     0.0
TEMPERATURE COEFFICIENTS
 1.05    1.07     1.07     1.07     1.07    1.07     1.05     1.05

PHOSPHORUS
SURFACE
 0.02      0.0      0.01     1.0     0.015
UPPER ZONE
 0.002    0.0      0.70     1.0     0.0015
LOWER ZONE
 0.002    0.0      0.80     1.0     0.005
GROUNDWATER
 0.0      0.0      0.0      0.0     0.0
TEMPERATURE COEFFICIENTS
 1.07     1.07     1.07     1.05     1.05
END
```

```
INITIAL
NITROGEN
SURFACE
 90.0     2.0     20.0    0.20    0.0     0.0
UPPER ZONE
 375.0    4.0     40.0     3.00   0.0     0.0
LOWER ZONE
2500.     6.0    20.0     212.    0.0     0.0
GROUNDWATER
  0.0     0.0     0.0     0.0     0.0     0.0

PHOSPHORUS
SURFACE
  12.0      1.0   36.0    0.0
UPPER ZONE
 100.    0.35    305.0    0.0
LOWER ZONE
 800.      1.0   135.0    0.0
GROUNDWATER
  0.0     0.0     0.0     0.0

CHLORIDE
SURFACE
  3.46
UPPER ZONE
  79.54
LOWER ZONE
  770.
GROUNDWATER
  0.0
END


APPLICATION  119
NITROGEN
SURFACE
     0.0   0.71    0.0     0.0     0.0     0.0
UPPER ZONE
     0.0  33.29     0.0    0.0     0.0     0.0

PHOSPHORUS
SURFACE
     0.0    0.2     0.40    0.0
UPPER ZONE
    0.0    28.9    0.0     0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END
```

```
APPLICATION  162
NITROGEN
SURFACE
    0.0     25.0    3.0      0.0      0.0      0.0
UPPER ZONE
   0.0     61.9    0.0     0.0     0.0     0.0

PHOSPHORUS
SURFACE
     0.0     0.0     0.0     0.0
UPPER ZONE
     0.0     0.0     0.0     0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END


APPLICATION  297
NITROGEN
SURFACE
    1.5     0.0     0.0      0.0      0.0      0.0
UPPER ZONE
  20.0      0.0    0.0     0.0     0.0     0.0

PHOSPHORUS
SURFACE
     0.3     0.0     0.0     0.0
UPPER ZONE
     4.0     0.0     0.0     0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END
 LZTEMP=  48.0  47.0  53.0  57.0  65.0  73.0  77.0  77.0  76.0  71.0  58.0  51.0
 ASZT=  -21.4  BSZT=   1.345 AUZT=  26.210 BUZT=   0.675
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ=99.90 UZF=55.0 LZF=1.0
```

```
P-2 WATKINSVILLE, GA.: SWCP - TERRACES (2) WITH CONTOURING
NUTRIENTS APPLIED: N & P ON DAY 119; N ON DAY 162; PLANT DECAY AFTER HARVEST
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=NO
PEST=NO
NUTR=YES
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=     3.20
 BGNDAY= 2 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.65 UZS=     0.650 LZSN=   18.00 LZS=     18.00
 L=     250. SS=  0.0150 NN=  0.2500 A=    0.0000 EPXM=0.1200 PETML 1.000
 K3=      0.300 0.300 0.300 0.400 0.400 0.500 0.700 0.800 0.600 0.500 0.400 0.300
 INFIL 0.100 INTER 0.590 IRC=  0.000 K24L= 1.000 KK24= 0.600 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
 COVPMO= 0.100 0.100 0.100 0.100 0.000 0.150 0.600 0.850 0.750 0.100 0.100 0.100
 TIMTIL= 115 115 115 115 115 115 115 115 115 115   0   0 SMPF=0.35
 YRTIL=   66  67  68  69  70  71  72  73  74  75  99  99
 SRERTL= 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 1.500 0.000 0.000
 JRER= 1.9  KRER= 0.160 JSER= 1.700 KSER= 0.500 SRERI= 2.00 SCMPAC 0.02
NUTRIENT
  TSTEP= 360  NAPPL=   3 TIMHAR= 297
 ULUPTK=0.00  0.05  0.08  0.10  0.15  0.75  0.200  0.07 0.00  0.00  0.00   0.0
 LZUPTK=0.00  0.00  0.00  0.05  0.03  0.55  0.450  0.10 0.00  0.00  0.00   0.0

REACTION RATES
NITROGEN
SURFACE
 1.00    0.0     0.10     0.00   0.0     0.0      1.0    .20
UPPER ZONE
 0.20    0.006    0.13    0.002   0.00     0.0     1.0     0.25
LOWER ZONE
 0.10    0.002    0.025    0.002   0.0      0.0      1.0     0.2
GROUNDWATER
 0.0     0.0     0.0     0.0     0.0     0.0      0.0     0.0
TEMPERATURE COEFFICIENTS
 1.05    1.07    1.07    1.07    1.07    1.07    1.05    1.05

PHOSPHORUS
SURFACE
 0.02     0.0     0.01    1.0    0.015
UPPER ZONE
 0.002   0.0     0.70    1.0     0.0015
LOWER ZONE
 0.002   0.0    0.80    1.0     0.005
GROUNDWATER
 0.0     0.0     0.0     0.0     0.0
TEMPERATURE COEFFICIENTS
 1.07    1.07    1.07    1.05    1.05
END
```

```
INITIAL
NITROGEN
SURFACE
 90.0     2.0     20.0    0.20    0.0     0.0
UPPER ZONE
 375.0    4.0     40.0     3.00   0.0     0.0
LOWER ZONE
2500.     6.0    20.0      212.   0.0     0.0
GROUNDWATER
  0.0     0.0     0.0     0.0     0.0     0.0

PHOSPHORUS
SURFACE
  12.0     1.0    36.0    0.0
UPPER ZONE
 100.    0.35    305.0    0.0
LOWER ZONE
 800.     1.0    135.0    0.0
GROUNDWATER
  0.0     0.0     0.0     0.0

CHLORIDE
SURFACE
  3.46
UPPER ZONE
  79.54
LOWER ZONE
  770.
GROUNDWATER
  0.0
END


APPLICATION  119
NITROGEN
SURFACE
     0.0    0.71    0.0      0.0     0.0     0.0
UPPER ZONE
     0.0   33.29     0.0     0.0     0.0     0.0

PHOSPHORUS
SURFACE
     0.0     0.2     0.40    0.0
UPPER ZONE
    0.0     28.9    0.0     0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END
```

```
APPLICATION  162
NITROGEN
SURFACE
    0.0     25.0    3.0      0.0      0.0      0.0
UPPER ZONE
   0.0     61.9    0.0      0.0      0.0      0.0

PHOSPHORUS
SURFACE
     0.0     0.0      0.0      0.0
UPPER ZONE
     0.0     0.0      0.0      0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END


APPLICATION  297
NITROGEN
SURFACE
    1.5     0.0      0.0      0.0      0.0      0.0
UPPER ZONE
  20.0      0.0    0.0      0.0      0.0      0.0

PHOSPHORUS
SURFACE
     0.3     0.0      0.0      0.0
UPPER ZONE
     4.0     0.0      0.0      0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END
 LZTEMP=  48.0  47.0  53.0  57.0  65.0  73.0  77.0  77.0  76.0  71.0  58.0  51.0
 ASZT=  -21.4  BSZT=   1.345 AUZT=  26.210 BUZT=   0.675
 SZDPTH=0.1250 UZDPTH= 6.125 BDSZ= 99.90 BDUZ= 99.90 BDLZ=99.90 UZF=5.0 LZF=1.0
```

P6 Parameter Values

Atrazine--

```
P-6 EAST LANSING, MI.: BASE CONDITIONS (NO SWCP)
ATRAZINE APPLIED: 2.5 LBS/AC EVERY DAY 140 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.170 UZS=    0.250 LZSN=    9.00 LZS=      7.0
 L=      60. SS=  0.0600 NN=  0.2000 A=   0.0000 EPXM=0.1200 PETML 0.700
 K3=     0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=    0.000
 DGM=  0.000 WC=   0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=   1.000 RMUL=   1.000 KUGI=   0.000
 COVPMO= 0.000 0.000 0.000 0.000 0.000 0.050 0.550 0.900 0.900 0.800 0.000 0.000
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.60 SRERI= 1.0  SCMPAC .001
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  140 140 140 140 140 140 140 140 140 140   0   0 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  00  00
 SSTR=   2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 0.000 0.000
 CMAX=  3.5E-5 DD=    0.0000 K=     4.0000 N=     1.0000 NP=    2.300
 DDG=    140 140 140 140 140 140 140 140 140 140   0   0
 YDG=     66  67  68  69  70  71  72  73  74  75  00  00
 KDG=    0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.0   0.0
 SZDPTH=0.1250 UZDPTH=6.1250 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=1.0 LZF=1.0
```

```
P-6 EAST LANSING, MI.: NO TILLAGE SWCP
ATRAZINE APPLIED: 2.5 LBS/AC EVERY DAY 140 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.170 UZS=     0.250 LZSN=    9.00 LZS=      7.0
 L=       60. SS=  0.0600 NN=  0.3200 A=   0.0000 EPXM=0.1200 PETML 0.700
 K3=      0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=    0.000
 DGM=  0.000 WC=   0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=   1.000 RMUL=   1.000 KUGI=   0.000
 COVPMO= 0.600 0.600 0.600 0.600 0.400 0.430 0.730 0.940 0.940 0.880 0.600 0.600
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.55 SRERI= 0.1  SCMPAC .001
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  140 140 140 140 140 140 140 140 140 140   0   0 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  00  00
 SSTR=   2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 0.000 0.000
 CMAX=  3.5E-5 DD=    0.0000 K=     4.0000 N=     1.0000 NP=    2.300
 DDG=    140 140 140 140 140 140 140 140 140 140   0   0
 YDG=     66  67  68  69  70  71  72  73  74  75  00  00
 KDG=    0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.0   0.0
 SZDPTH=0.1250 UZDPTH=6.1250 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=1.0 LZF=1.0
```

```
P-6 EAST LANSING, MI.: CONTOUR SWCP
ATRAZINE APPLIED: 2.5 LBS/AC EVERY DAY 140 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.200 UZS=    0.300 LZSN=    9.00 LZS=      7.0
 L=      60. SS=  0.0600 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 0.700
 K3=     0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.80C IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=    0.000
 DGM=  0.000 WC=   0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=   1.000 RMUL=   1.000 KUGI=   0.000
 COVPMO= 0.000 0.000 0.000 0.000 0.000 0.050 0.550 0.900 0.900 0.800 0.000 0.000
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=0.5
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.50 SRERI= 1.0  SCMPAC .001
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  140 140 140 140 140 140 140 140 140 140   0   0 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  00  00
 SSTR=   2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 0.000 0.000
 CMAX=  3.5E-5 DD=     0.0000 K=     4.0000 N=     1.0000 NP=    2.300
 DDG=    140 140 140 140 140 140 140 140 140 140   0   0
 YDG=     66  67  68  69  70  71  72  73  74  75  00  00
 KDG=    0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.0   0.0
 SZDPTH=0.1250 UZDPTH=6.1250 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=1.0 LZF=1.0
```

```
P-6 EAST LANSING, MI.: TERRACE WITH CONTOURS SWCP
ATRAZINE APPLIED: 2.5 LBS/AC EVERY DAY 140 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=     1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.260 UZS=     0.390 LZSN=    9.00 LZS=       7.0
 L=      200. SS=  0.0180 NN=  0.2500 A=    0.0000 EPXM=0.1200 PETML 0.700
 K3=     0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=    0.000
 DGM=  0.000 WC=   0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=   1.000 RMUL=   1.000 KUGI=   0.000
 COVPMO= 0.000 0.000 0.000 0.000 0.000 0.050 0.550 0.900 0.900 0.800 0.000 0.000
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=0.35
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.50 SRERI= 1.0  SCMPAC .001
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  140 140 140 140 140 140 140 140 140 140   0   0 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  00  00
 SSTR=   2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 2.500 0.000 0.000
 CMAX=  3.5E-5 DD=     0.0000 K=      4.0000 N=      1.0000 NP=    2.300
 DDG=    140 140 140 140 140 140 140 140 140 140   0   0
 YDG=     66  67  68  69  70  71  72  73  74  75  00  00
 KDG=    0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.050 0.0   0.0
 SZDPTH=0.1250 UZDPTH=6.1250 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=1.0 LZF=1.0
```

Paraquat--

```
P-6 EAST LANSING, MI.: BASE CONDITIONS (NO SWCP)
PARAQUAT APPLIED: 1.3 LBS/AC EVERY DAY 140 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.170 UZS=    0.250 LZSN=    9.00 LZS=      7.0
 L=      60. SS=  0.0600 NN=  0.2000 A=   0.0000 EPXM=0.1200 PETML 0.700
 K3=     0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=    0.000
 DGM=  0.000 WC=   0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=   1.000 RMUL=   1.000 KUGI=   0.000
 COVPMO= 0.000 0.000 0.000 0.000 0.000 0.050 0.550 0.900 0.900 0.800 0.000 0.000
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.60 SRERI= 1.0  SCMPAC .001
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  140 140 140 140 140 140 140 140 140 140   0   0 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  00  00
 SSTR=   1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 0.000 0.000
 CMAX=  1.0E-5 DD=    0.0003 K=     120.00 N=     2.0000 NP=    4.600
 DDG=    140 140 140 140 140 140 140 140 140 140   0   0
 YDG=     66  67  68  69  70  71  72  73  74  75  00  00
 KDG=    0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.0   0.0
 SZDPTH=0.1250 UZDPTH=6.1250 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=1.0 LZF=1.0
```

```
P-6 EAST LANSING, MI.: NO TILLAGE SWCP
PARAQUAT APPLIED: 1.3 LBS/AC EVERY DAY 140 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.170 UZS=    0.250 LZSN=    9.00 LZS=      7.0
 L=        60. SS=  0.0600 NN=  0.3200 A=   0.0000 EPXM=0.1200 PETML 0.700
 K3=      0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=    0.000
 DGM=  0.000 WC=   0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=   1.000 RMUL=   1.000 KUGI=   0.000
 COVPMO= 0.600 0.600 0.600 0.600 0.400 0.430 0.730 0.940 0.940 0.880 0.600 0.600
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.55 SRERI= 0.1  SCMPAC .001
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  140 140 140 140 140 140 140 140 140 140   0   0 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  00  00
 SSTR=   1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 0.000 0.000
 CMAX=  1.0E-5 DD=     0.0003 K=      120.00 N=      2.0000 NP=     4.600
 DDG=    140 140 140 140 140 140 140 140 140 140   0   0
 YDG=     66  67  68  69  70  71  72  73  74  75  00  00
 KDG=    0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.0   0.0
 SZDPTH=0.1250 UZDPTH=6.1250 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=1.0 LZF=1.0
```

```
P-6 EAST LANSING, MI.: CONTOUR SWCP
PARAQUAT APPLIED: 1.3 LBS/AC EVERY DAY 140 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=     1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.200 UZS=     0.300 LZSN=     9.00 LZS=      7.0
 L=       60. SS=  0.0600 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 0.700
 K3=      0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=    0.000
 DGM=  0.000 WC=   0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=   1.000 RMUL=   1.000 KUGI=   0.000
 COVPMO= 0.000 0.000 0.000 0.000 0.000 0.050 0.550 0.900 0.900 0.800 0.000 0.000
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=0.5
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.50 SRERI= 1.0  SCMPAC .001
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  140 140 140 140 140 140 140 140 140 140   0   0 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  00  00
 SSTR=   1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 0.000 0.000
 CMAX=  1.0E-5 DD=     0.0003 K=      120.00 N=      2.0000 NP=    4.600
 DDG=     140 140 140 140 140 140 140 140 140 140   0   0
 YDG=      66  67  68  69  70  71  72  73  74  75  00  00
 KDG=     0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.0   0.0
 SZDPTH=0.1250 UZDPTH=6.1250 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=1.0 LZF=1.0
```

```
P-6 EAST LANSING, MI.: TERRACE WITH CONTOURS SWCP
PARAQUAT APPLIED: 1.3 LBS/AC EVERY DAY 140 FOR THE 10-YR PERIOD
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=YES
NUTR=NO
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.260 UZS=    0.390 LZSN=    9.00 LZS=      7.0
 L=     200. SS=  0.0180 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 0.700
 K3=     0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=    0.000
 DGM=  0.000 WC=   0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=   1.000 RMUL=   1.000 KUGI=   0.000
 COVPMO= 0.000 0.000 0.000 0.000 0.000 0.050 0.550 0.900 0.900 0.800 0.000 0.000
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=0.35
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.50 SRERI= 1.0  SCMPAC .001
PESTICIDE
APMODE=SURF
DESORP=YES
 PSSZ=   0.00  PSUZ=   0.00  PSLZ=   0.00  PSGZ=   0.0
 TIMAP=  140 140 140 140 140 140 140 140 140 140   0   0 NPA= 10 NDRPA=  1
 YEARAP=  66  67  68  69  70  71  72  73  74  75  00  00
 SSTR=   1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 1.300 0.000 0.000
 CMAX=  1.0E-5 DD=    0.0003 K=     120.00 N=      2.0000 NP=    4.600
 DDG=    140 140 140 140 140 140 140 140 140 140   0   0
 YDG=     66  67  68  69  70  71  72  73  74  75  00  00
 KDG=    0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.002 0.0   0.0
 SZDPTH=0.1250 UZDPTH=6.1250 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=1.0 LZF=1.0
```

Nutrients--

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P-6 EAST LANSING, MI.: BASE CONDITIONS (NO SWCP)
NUTRIENTS APPLIED: N & P ON DAY 140; N ON DAY 189; PLANT DECAY AFTER HARVEST
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=NO
NUTR=YES
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=    1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.170 UZS=    0.250 LZSN=    9.00 LZS=      7.0
 L=      60. SS=  0.0600 NN=  0.2000 A=   0.0000 EPXM=0.1200 PETML 0.700
 K3=     0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=    0.000
 DGM=  0.000 WC=   0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=   1.000 RMUL=   1.000 KUGI=   0.000
 COVPMO= 0.000 0.000 0.000 0.000 0.000 0.050 0.550 0.900 0.900 0.800 0.000 0.000
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 1.000 1.000 1.000 1.000 1.00C 1.000 1.000 1.000 1.000 1.000 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.60 SRERI= 1.0  SCMPAC .001
NUTRIENT
  TSTEP= 360  NAPPL=   3 TIMHAR= 273
 ULUPTK=0.00  0.00  0.00  0.01  0.05  0.09  1.000 0.060 0.00  0.00  0.00   0.0
 LZUPTK=0.00  0.00  0.00  0.00  0.00  0.01  1.000 0.125 0.00  0.00  0.00   0.0

REACTION RATES
NITROGEN
SURFACE
  3.00   0.0      0.25     0.015     0.0      0.0      5.0     0.75
UPPER ZONE
  1.25    0.05     0.40    0.0015    0.0      0.0     0.75      0.3
LOWER ZONE
   0.7    0.0     0.090   0.0015     0.0      0.0      1.0      0.4
GROUNDWATER
   0.0    0.0      0.0     0.0       0.0      0.0      0.0      0.0
TEMPERATURE COEFFICIENTS
  1.05    1.07     1.07     1.07     1.07     1.07     1.05     1.05

PHOSPHORUS
SURFACE
 0.015   0.0      0.01     1.00      0.01
UPPER ZONE
 0.0015  0.0      2.10     0.5      0.006
LOWER ZONE
 0.0015  0.0      1.70     0.5      0.005
```

```
GROUNDWATER
   0.0        0.0    0.0    0.0     0.0
TEMPERATURE COEFFICIENTS
   1.07    1.07    1.07   1.05    1.05
END

INITIAL
NITROGEN
SURFACE
  68.00   0.40    1.00    1.95    0.0     0.0
UPPER ZONE
 510.0     8.00   20.50   27.4    0.0     0.0
LOWER ZONE
  2087.    9.0    23.0    100.0   0.0     0.0
GROUNDWATER
  0.0     0.0     0.0     0.0     0.0     0.0

PHOSPHORUS
SURFACE
 27.3     0.09    6.64    0.0
UPPER ZONE
  175.     13.3    101.3   0.0
LOWER ZONE
 1000.     26.8   241.5     0.0
GROUNDWATER
  0.0     0.0     0.0     0.0

CHLORIDE
SURFACE
  6.65
UPPER ZONE
 108.0
LOWER ZONE
  147.6
GROUNDWATER
  0.0
END


APPLICATION  140
NITROGEN
SURFACE
     0.0    0.3    1.0     1.3      0.0     0.0
UPPER ZONE
     0.0    29.2    0.0   29.2     0.0     0.0

PHOSPHORUS
SURFACE
     0.0    0.35    3.10     0.0
UPPER ZONE
     0.0    79.5     0.0     0.0

CHLORIDE
SURFACE
```

```
     5.8
UPPER ZONE
  134.2
END


APPLICATION  189
NITROGEN
SURFACE
     0.0    38.5     1.5     40.0     0.0      0.0
UPPER ZONE
     0.0     18.0    0.0      18.0    0.0      0.0

PHOSPHORUS
SURFACE
     0.0     0.0     0.0      0.0
UPPER ZONE
     0.0     0.0     0.0      0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END

APPLICATION  273
NITROGEN
SURFACE
     1.5     0.0     0.0      0.0     0.0      0.0
UPPER ZONE
    16.0      0.0    0.0       0.0    0.0      0.0

PHOSPHORUS
SURFACE
    0.25     0.0     0.0      0.0
UPPER ZONE
     3.0     0.0     0.0      0.0

CHLORIDE
SURFACE
   66.0
UPPER ZONE
  60.0
END
 LZTEMP=  38.2  36.6  37.1  40.1  48.5  56.5  62.4  65.1  64.5  58.7  51.33  44.3
 ASZT=  44.96  BSZT=   0.39  AUZT=   0.0   BUZT=   1.0
 SZDPTH=0.1250 UZDPTH= 3.125 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=5.0 LZZF=1.0
```

```
P-6 EAST LANSING, MI.: NO TILLAGE SWCP
NUTRIENTS APPLIED: N & P ON DAY 140; N ON DAY 189; PLANT DECAY AFTER HARVEST
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=NO
NUTR=YES
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=     1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.170 UZS=     0.250 LZSN=     9.00 LZS=       7.0
 L=       60. SS=  0:0600 NN=  0.3200 A=   0.0000 EPXM=0.1200 PETML 0.700
 K3=      0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=    0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=     0.000
 DGM=  0.000 WC=    0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=    1.000 RMUL=   1.000 KUGI=    0.000
 COVPMO= 0.600 0.600 0.600 0.600 0.400 0.430 0.730 0.940 0.940 0.880 0.600 0.600
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=1.0
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.100 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.55 SRERI= 0.1  SCMPAC .001
NUTRIENT
  TSTEP= 360  NAPPL=   3 TIMHAR= 273
 ULUPTK=0.00  0.00  0.00  0.01  0.05  0.09  1.000 0.060 0.00  0.00  0.00   0.0
 LZUPTK=0.00  0.00  0.00  0.00  0.00  0.01  1.000 0.125 0.00  0.00  0.00   0.0

REACTION RATES
NITROGEN
SURFACE
  3.00   0.0     0.25    0.015    0.0      0.0      5.0     0.75
UPPER ZONE
  1.25   0.05    0.40    0.0015   0.0      0.0     0.75      0.3
LOWER ZONE
   0.7    0.0    0.090   0.0015    0.0      0.0     1.0      0.4
GROUNDWATER
   0.0    0.0     0.0     0.0       0.0      0.0     0.0      0.0
TEMPERATURE COEFFICIENTS
  1.05   1.07    1.07     1.07     1.07     1.07     1.05     1.05

PHOSPHORUS
SURFACE
 0.015   0.0     0.01    1.00     0.01
UPPER ZONE
 0.0015  0.0     2.10    0.5     0.006
LOWER ZONE
 0.0015  0.0     1.70    0.5     0.005
```

```
GROUNDWATER
   0.0       0.0    0.0    0.0     0.0
TEMPERATURE COEFFICIENTS
   1.07   1.07   1.07   1.05   1.05
END

INITIAL
NITROGEN
SURFACE
  68.00   0.40   1.00   1.95   0.0    0.0
UPPER ZONE
 510.0    8.00   20.50  27.4   0.0    0.0
LOWER ZONE
  2087.   9.0    23.0   100.0  0.0    0.0
GROUNDWATER
  0.0    0.0    0.0    0.0    0.0    0.0

PHOSPHORUS
SURFACE
 27.3     0.09   6.64   0.0
UPPER ZONE
  175.    13.3   101.3  0.0
LOWER ZONE
 1000.    26.8   241.5   0.0
GROUNDWATER
  0.0    0.0    0.0    0.0

CHLORIDE
SURFACE
  6.65
UPPER ZONE
 108.0
LOWER ZONE
  147.6
GROUNDWATER
  0.0
END


APPLICATION  140
NITROGEN
SURFACE
     0.0   0.3    1.0    1.3     0.0    0.0
UPPER ZONE
     0.0   29.2   0.0   29.2    0.0    0.0

PHOSPHORUS
SURFACE
     0.0   0.35   3.10    0.0
UPPER ZONE
     0.0   79.5    0.0    0.0

CHLORIDE
SURFACE
```

```
     5.8
UPPER ZONE
  134.2
END


APPLICATION  189
NITROGEN
SURFACE
     0.0    38.5     1.5    40.0     0.0      0.0
UPPER ZONE
     0.0     18.0    0.0     18.0    0.0      0.0

PHOSPHORUS
SURFACE
     0.0     0.0     0.0     0.0
UPPER ZONE
     0.0     0.0     0.0     0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END

APPLICATION  273
NITROGEN
SURFACE
     6.0     0.0     0.0     0.0     0.0      0.0
UPPER ZONE
    16.0      0.0    0.0      0.0    0.0      0.0

PHOSPHORUS
SURFACE
    1.20     0.0     0.0     0.0
UPPER ZONE
     3.0     0.0     0.0     0.0

CHLORIDE
SURFACE
   66.0
UPPER ZONE
  60.0
END
 LZTEMP=  38.2  36.6  37.1  40.1  48.5  56.5  62.4  65.1  64.5  58.7  51.3  44.3
 ASZT=  44.96  BSZT=   0.39  AUZT=   0.0   BUZT=   1.0
 SZDPTH=0.1250 UZDPTH= 3.125 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=5.0 LZF=1.0
```

```
P-6 EAST LANSING, MI.: CONTOUR SWCP
NUTRIENTS APPLIED: N & P ON DAY 140; N ON DAY 189; PLANT DECAY AFTER HARVEST
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=NO
NUTR=YES
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=     1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.200 UZS=     0.300 LZSN=     9.00 LZS=       7.0
 L=       60. SS=  0.0600 NN=  0.2500 A=    0.0000 EPXM=0.1200 PETML 0.700
 K3=      0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=    0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=     0.000
 DGM=  0.000 WC=    0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=    1.000 RMUL=    1.000 KUGI=    0.000
 COVPMO= 0.000 0.000 0.000 0.000 0.000 0.050 0.550 0.900 0.900 0.800 0.000 0.000
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=0.5
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.50 SRERI= 1.0  SCMPAC .001
NUTRIENT
  TSTEP= 360  NAPPL=    3 TIMHAR= 273
 ULUPTK=0.00  0.00  0.00  0.01  0.05  0.09  1.000 0.060 0.00  0.00  0.00   0.0
 LZUPTK=0.00  0.00  0.00  0.00  0.00  0.01  1.000 0.125 0.00  0.00  0.00   0.0

REACTION RATES
NITROGEN
SURFACE
  3.00  0.0     0.25     0.015    0.0      0.0      5.0     0.75
UPPER ZONE
  1.25   0.05    0.40    0.0015    0.0      0.0     0.75      0.3
LOWER ZONE
   0.7    0.0    0.090   0.0015     0.0      0.0     1.0      0.4
GROUNDWATER
   0.0    0.0     0.0     0.0       0.0      0.0      0.0      0.0
TEMPERATURE COEFFICIENTS
  1.05   1.07    1.07     1.07     1.07     1.07     1.05     1.05

PHOSPHORUS
SURFACE
 0.015  0.0     0.01     1.00      0.01
UPPER ZONE
 0.0015 0.0     2.10     0.5      0.006
LOWER ZONE
 0.0015 0.0     1.70     0.5      0.005
```

```
GROUNDWATER
   0.0         0.0    0.0    0.0     0.0
TEMPERATURE COEFFICIENTS
   1.07    1.07    1.07   1.05    1.05
END

INITIAL
NITROGEN
SURFACE
  68.00   0.40    1.00    1.95    0.0     0.0
UPPER ZONE
 510.0     8.00   20.50   27.4    0.0     0.0
LOWER ZONE
  2087.    9.0    23.0    100.0   0.0     0.0
GROUNDWATER
  0.0     0.0     0.0     0.0     0.0     0.0

PHOSPHORUS
SURFACE
 27.3     0.09    6.64    0.0
UPPER ZONE
  175.     13.3    101.3   0.0
LOWER ZONE
 1000.     26.8   241.5    0.0
GROUNDWATER
  0.0     0.0     0.0     0.0

CHLORIDE
SURFACE
  6.65
UPPER ZONE
 108.0
LOWER ZONE
  147.6
GROUNDWATER
  0.0
END


APPLICATION  140
NITROGEN
SURFACE
     0.0    0.3    1.0     1.3       0.0     0.0
UPPER ZONE
     0.0    29.2    0.0   29.2      0.0     0.0

PHOSPHORUS
SURFACE
     0.0    0.35    3.10     0.0
UPPER ZONE
     0.0    79.5     0.0     0.0

CHLORIDE
SURFACE
```

```
     5.8
UPPER ZONE
   134.2
END


APPLICATION  189
NITROGEN
SURFACE
     0.0    38.5     1.5     40.0     0.0     0.0
UPPER ZONE
     0.0    18.0     0.0     18.0     0.0     0.0

PHOSPHORUS
SURFACE
     0.0     0.0     0.0      0.0
UPPER ZONE
     0.0     0.0     0.0      0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END

APPLICATION  273
NITROGEN
SURFACE
     1.5     0.0     0.0     0.0     0.0     0.0
UPPER ZONE
    16.0     0.0     0.0     0.0     0.0     0.0

PHOSPHORUS
SURFACE
    0.25     0.0     0.0     0.0
UPPER ZONE
     3.0     0.0     0.0     0.0

CHLORIDE
SURFACE
   66.0
UPPER ZONE
  60.0
END
 LZTEMP=  38.2  36.6  37.1  40.1  48.5  56.5  62.4  65.1  64.5  58.7  51.3  44.3
 ASZT=  44.96  BSZT=   0.39  AUZT=   0.0   BUZT=   1.0
 SZDPTH=0.1250 UZDPTH= 3.125 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=5.0 LZF=1.0
```

```
P-6 EAST LANSING, MI.: TERRACE WITH CONTOURS SWCP
NUTRIENTS APPLIED: N & P ON DAY 140; N ON DAY 189; PLANT DECAY AFTER HARVEST
HYCAL=CALB
INPUT=ENGL
OUTPUT=ENGL
PRINT=INTR
SNOW=YES
PEST=NO
NUTR=YES
ICHECK=OFF
DISK=NO
 INTRVL=15 HYMIN=  0.100 AREA=     1.98
 BGNDAY= 7 BGNMON= 1 BGNYR= 1966
 ENDDAY=31 ENDMON=12 ENDYR= 1975
 UZSN=   0.260 UZS=    0.390 LZSN=    9.00 LZS=      7.0
 L=     200. SS=  0.0180 NN=  0.2500 A=   0.0000 EPXM=0.1200 PETML 0.700
 K3=     0.200 0.200 0.200 0.200 0.300 0.300 0.500 0.450 0.400 0.300 0.200 0.200
 INFIL 0.030 INTER 0.800 IRC=  0.000 K24L= 1.000 KK24= 0.000 K24EL 0.000
 SGW=  0.000 GWS=  0.000 KV=   0.000 ICS=  0.000 OFS=  0.000 IFS=  0.000
SNOWPRINT=NO
 RADCN 1.000 CCFAC 1.000 SCF=  1.400 ELDIF 0.000 IDNS= 0.140 F=    0.000
 DGM=  0.000 WC=   0.030 MPACK 1.000 EVAPS 0.400 MELEV  892. TSNOW 32.00
 PACK=   1.000 DEPTH=  4.000
 PETMIN= 35.00 PETMAX= 40.00 WMUL=   1.000 RMUL=   1.000 KUGI=   0.000
 COVPMO= 0.000 0.000 0.000 0.000 0.000 0.050 0.550 0.900 0.900 0.800 0.000 0.000
 TIMTIL= 139 139 139 139 139 139 139 139 139 139   0   0 SMPF=0.35
 YRTIL=   66  67  68  69  70  71  72  73  74  75   0   0
 SRERTL= 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 0.000 0.000
 JRER= 2.200 KRER= 0.150 JSER= 1.400 KSER= 0.50 SRERI= 1.0  SCMPAC .001
NUTRIENT
  TSTEP= 360  NAPPL=   3 TIMHAR= 273
 ULUPTK=0.00  0.00  0.00  0.01  0.05  0.09  1.000 0.060 0.00  0.00  0.00   0.0
 LZUPTK=0.00  0.00  0.00  0.00  0.00  0.01  1.000 0.125 0.00  0.00  0.00   0.0

REACTION RATES
NITROGEN
SURFACE
  3.00   0.0     0.25    0.015    0.0     0.0     5.0     0.75
UPPER ZONE
  1.25    0.05    0.40   0.0015   0.0     0.0    0.75      0.3
LOWER ZONE
   0.7    0.0    0.090   0.0015    0.0     0.0    1.0     0.4
GROUNDWATER
   0.0    0.0     0.0    0.0      0.0     0.0     0.0     0.0
TEMPERATURE COEFFICIENTS
  1.05    1.07    1.07    1.07    1.07    1.07    1.05    1.05

PHOSPHORUS
SURFACE
 0.015   0.0     0.01    1.00     0.01
UPPER ZONE
 0.0015  0.0     2.10    0.5     0.006
LOWER ZONE
 0.0015  0.0     1.70    0.5     0.005
```

```
GROUNDWATER
   0.0       0.0    0.0    0.0     0.0
TEMPERATURE COEFFICIENTS
   1.07    1.07    1.07   1.05    1.05
END

INITIAL
NITROGEN
SURFACE
  68.00   0.40    1.00    1.95    0.0     0.0
UPPER ZONE
 510.0    8.00    20.50   27.4    0.0     0.0
LOWER ZONE
  2087.   9.0     23.0    100.0   0.0     0.0
GROUNDWATER
  0.0     0.0     0.0     0.0     0.0     0.0

PHOSPHORUS
SURFACE
 27.3     0.09    6.64    0.0
UPPER ZONE
  175.     13.3    101.3   0.0
LOWER ZONE
 1000.     26.8   241.5     0.0
GROUNDWATER
  0.0     0.0     0.0     0.0

CHLORIDE
SURFACE
  6.65
UPPER ZONE
 108.0
LOWER ZONE
  147.6
GROUNDWATER
  0.0
END


APPLICATION  140
NITROGEN
SURFACE
     0.0    0.3    1.0     1.3      0.0      0.0
UPPER ZONE
     0.0    29.2    0.0   29.2     0.0      0.0

PHOSPHORUS
SURFACE
     0.0    0.35    3.10     0.0
UPPER ZONE
     0.0    79.5     0.0     0.0

CHLORIDE
SURFACE
```

```
     5.8
UPPER ZONE
   134.2
END


APPLICATION  189
NITROGEN
SURFACE
     0.0    38.5     1.5    40.0     0.0     0.0
UPPER ZONE
     0.0     18.0    0.0     18.0    0.0     0.0

PHOSPHORUS
SURFACE
     0.0     0.0     0.0     0.0
UPPER ZONE
     0.0     0.0     0.0     0.0

CHLORIDE
SURFACE
     0.0
UPPER ZONE
     0.0
END

APPLICATION  273
NITROGEN
SURFACE
     1.5     0.0     0.0     0.0     0.0     0.0
UPPER ZONE
    16.0      0.0    0.0      0.0    0.0     0.0

PHOSPHORUS
SURFACE
    0.25     0.0     0.0     0.0
UPPER ZONE
     3.0     0.0     0.0     0.0

CHLORIDE
SURFACE
    66.0
UPPER ZONE
   60.0
END
 LZTEMP=  38.2  36.6  37.1  40.1  48.5  56.5  62.4  65.1  64.5  58.7  51.3  44.3
 ASZT=  44.96  BSZT=   0.39  AUZT=   0.0   BUZT=   1.0
 SZDPTH=0.1250 UZDPTH= 3.125 BDSZ= 63.70 BDUZ= 72.40 BDLZ= 99.0 UZF=5.0 LZF=1.0
```